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Dr. Peter F. Ffolliott
School of Natural Resources and the Environment
Bio Sciences East 220 (Building 43)
P.O. Box 210043
University of Arizona
Tucson, Arizona
(520)621-7276
ffolpete@ag.arizona.edu

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FIRE EFFECTS ON SOIL EROSION AND DEPOSITION ON HILLSLOPES IN THE OAK SAVANNAS

Peter F. Ffolliott,[1] Gerald J. Gottfried,[2] Aaron T. Kauffman,[1] Cody L. Stropki,[1] and Hui Chen[1]

Fire was a natural occurrence in many ecosystems of the Southwest Borderlands before Euro-American settlement. However, natural fire frequencies and the burning characteristics of the fires have been altered since the early 1900s, largely because of past livestock grazing practices, which removed significant portions of the fire-carrying herbaceous vegetation, and past fire suppression policies of land management agencies (Edminster et al. 2000, Fulé and Covington 1995). These practices and policies have resulted in over-crowding of tree overstories on many sites, with the trees more susceptible to insects, disease, and wildfire. Mesquite (*Prosopis* spp.) and other woody plants have invaded otherwise productive grassland ecosystems in many instances. As a consequence of these conditions, land management agencies, with support from collaborating universities, private-sector organizations, and local stakeholders, are exploring the possibilities of re-introducing "more natural" fire regimes into many of the ecosystems in this region. Included among these ecosystems are the oak savannas, a plant community situated between the higher-elevation and more densely stocked oak woodlands and lower-elevation desert grassland-shrub vegetation.

Pre-fire estimates of soil erosion and deposition and an initial analysis of the effects of prescribed burning treatments and a wildfire on soil erosion and deposition in the oak savannas of the Malpai Borderlands are presented in this paper. This information should be useful in preparing strategies for re-introduction of "more natural" fire regimes into this region.

CASCABEL WATERSHEDS

Twelve small watersheds, ranging from 20 to almost 60 acres in size, in the Peloncillo Mountains of southwestern New Mexico (Gottfried et al. 2000, Gottfried and Edminster 2005) were (collectively) the study areas. The areal aggregation of these watersheds, called the Cascabel Watersheds, is 451.3 acres. The watersheds are situated between 5,380 and 5,590 feet in elevation. The nearest long-term precipitation station indicates that annual precipitation averages 23.5 inches, with nearly one-half occurring in summer rainfall. Streamflow originating on the watersheds is intermittent. However, large flows can be generated by high-intensity rainfall events (Gottfried et al. 2006). Vegetative, geologic, physiologic, and hydrologic characteristics of the watersheds are described elsewhere (Ffolliott and Gottfried 2005, Ffolliott et al. 2008, Gottfried et al. 2000, 2007, Neary and Gottfried 2004, Osterkamp 1999, Robertson et al. 2002, Vincent 1998, and Youberg and Ferguson 2001) and, therefore, not presented in this paper.

PRESCRIBED FIRE TREATMENTS AND WILDFIRE

The original objective of the research program on the Cascabel Watersheds was to evaluate the effects of warm-season (May through October) and cool-season (November through April) prescribed burning treatments on the natural resources including the tree overstories, herbaceous understories, wildlife populations and their habitats, erosion and deposition on the hillslopes and consequent sedimentation, and stormflow regimes. These evaluations would then be compared to unburned (control) watersheds to assess the impacts of the burning treatments. Following the required calibration period, four of the watersheds were burned during the cool-season in early March 2008. Three of the four watersheds to be burned in the warm-season were successfully burned on May 20, 2008, with burning of the fourth watershed scheduled for a later date. However, wind gusts up to 60 mph blew firebrands (burning embers) onto the remaining watershed to be burned and the four control watersheds in the morning of May 21, 2008. The resulting wildfire, designed the Whitmire Wildfire,

1 School of Natural Resources, University of Arizona, Tucson, Arizona
2 Rocky Mountain Research Station, U.S. Forest Service, Phoenix, Arizona

crossed the boundary lines among the watersheds and spread outward to burn approximately 4,000 acres.

The original objective of the research program on the watersheds had to be modified as a consequence of the wildfire. The current program is to accomplish the original objectives by evaluating the effects of cool-season and warm-season prescribed burning treatments and wildfire on the natural resources considered originally.

CLASSIFICATION OF FIRE SEVERITY

A system that relates fire severity to the soil-resource response to burning (Hungerford 1996) was the basis for classifying severity of the cool-season and warm-season prescribed burning treatments and wildfire at the sample plots located on the watersheds (see below) following the tree burning events. Classifications of fire severity at the sample plots were then extrapolated to a watershed-basis to determine the respective percentage of the Cascabel Watersheds that were burned at low, moderate, and high fired severities by the prescribed burns and wildfire.

These extrapolations indicated that 85 percent of the four watersheds experiencing the cool-season prescribed burn had been exposed to a low severity fire, a moderate fire severity was observed on 5 percent, and the remaining 10 percent were unburned (Stropki et al. 2009). Distributions of fire severities on the three watersheds exposed to the warm-season prescribed burn were similar generally to those of the cool-season burn. Three-fourths of the five watersheds burned by the wildfire had been exposed to a low severity fire, 20 percent to a moderate fire severity, and no burning effects were observed on 5 percent of the watersheds. It was concluded, therefore, that the Cascabel Watersheds burned by the prescribed treatments and wildfire had all been exposed to burning events of low severities.

STUDY PROTOCOLS

Sampling Basis

On each of the Cascabel Watersheds, between 35 and 45 sample plots had been located along transects perpendicular to the main stream systems and situated from ridge to ridge to provide the sampling basis for obtaining pre-burning data sets on the natural resources to be evaluated. The interval between the plots varied depending on the size and configuration of the watershed sampled. A total of 421 plots were located on the 12 watersheds. This basic sampling design had been used in other (earlier) studies of the natural resources on the Cascabel Watersheds (including Ffolliott and Gottfried 2005, Ffolliott et al. 2005, 2008, Gottfried et al. 2007, Jones et al. 2005, and Stropki et al. 2009).

Measurements of Soil Erosion and Deposition

Three capped pins were placed around every third plot on the watersheds to measure soil erosion and deposition. Two pins were placed 3 feet upslope and one pin 3 feet downslope of a plot. Seasonal measurements of soil loss beneath the cap of the pins (soil erosion) or soil accumulation on the cap if the pins (soil deposition) were made in the spring and fall to characterize soil movements following periods of winter rains and summer rainfalls, respectively. Occasionally, there was no measurable change in the soil surface beneath the cap. It was assumed in these cases that either the magnitudes of soil erosion and deposition in the time interval between the measurements equaled each other or (what is less likely) neither erosion nor deposition occurred in the time interval. The capped pins were re-set flush to the soil surface after each measurement to facilitate the subsequent measurements. Measurements obtained from the three pins surrounding a plot were averaged to estimate soil movement at the plot, with the plots on a watershed then averaged to described soil erosion and deposition on a watershed-basis.

Initial measurements were made in the fall of 2004 (Ffolliott et al. 2005) with pre-fire measurements continuing on a biannual basis until the burning events on the Cascabel Watersheds occurred. Post-fire measurements were made initially in the spring of 2008 shortly following the cool-season burns, with post-fire measurements of all three burning events made in the fall of 2008 and spring and fall of 2009. A bulk density value of 70.5 pounds/cubic-foot was used in converting the measurements of soil erosion and deposition to corresponding measures of tons per acre on a watershed-basis.

Measurements of soil erosion and deposition were analyzed separately as they represented separate processes of soil movements on the hillslopes of the watersheds. Because the frequency distributions of the two data sets were non-normal in structure and transformations of the data sets failed to normalize the distributions, the respective measurements of soil erosion and deposition were analyzed by "distribution-free" non-parametric tests, namely the Mann-Whitney and Nemenyi tests. These tests are outlined in Zar (1999) and other references on non-parametric methods of statistical analyses. Plots with no measurable change in soil movement were excluded from the analysis. All of the comparisons were evaluated at a 0.10 level of significance.

Other Measurements

Determining whether soil erosion or deposition might be related to precipitation, physiography, and/or vegetation on the watersheds was a study component. Rainfall amounts recorded at two weather stations located on the watersheds were averaged for each measurement period to determined if these seasonal amounts related to soil movement. Rainfall intensities were also determined from these records. High-intensity rainfall events can result in detachment of soil particles and subsequent downslope movement of the particles. However, the detachment of soil particles by the impact of raindrops is not necessarily a "proxy" of soil movement alone. Consideration of overland flows of water is also required (Parsons et al. 1999). Following a review of studies on the characteristics of rainfall events and resulting overland flows of water, Syed et al. (2003) selected a rainfall intensity of 0.25 inches/hours as a "minimum threshold" for generating overland flows in the region. This threshold was selected for the analysis of soil erosion and deposition on the Cascabel Watersheds in this study.

Rainfall amounts inducing overland flows of water was also considered as a possible variable related to soil movement. Osborn and Lane (1969) reported that individual rainfall events totaling 0.32 inches resulted in overland flows of water on the Lucky Hills Watershed near Tombstone, Arizona. Because of the "shallow" soils characterizing the Cascabel Watersheds, rainfall events of 0.35 inches or more were selected (arbitrarily) as the amount of rainfall assumed necessary to produce overland flows of water in this study.

High levels of antecedent soil moisture from prior rainfall events can lead to a situation where the infiltration capacity of the soil is exceeded with subsequent rainfall events. Syed et al. (2003) concluded that the antecedent soil moisture in the upper "few" inches of the surface is essentially "lost" within five days of a rainfall event. A more "conservative" estimate to two successive rainfall events of 0.35 inches or more occurring within three days of each other was selected arbitrarily as a "catalyst" for generating overland flows of water in this study.

Measured physiographic characteristics of the watersheds included hillslope position (upper, middle, and lower), hillslope steepness (to nearest 5 percent) and slope aspect (N, NE, E, etc.). An index integrating slope percent and aspect into values of potential solar beam radiation irradiation (Frank and Lee 1966) was analyzed to determined whether combinations of slope gradient and direction of a slope were related to soil erosion and deposition.

The numbers of trees were tallied on 1/4-acre circular plots (Ffolliott et al. 2008) and the crown closure of trees was measured above each of the plots with a 60° spherical densiometer (Lemmon 1956) to characterize the overstories. Annual production (standing biomass) of herbaceous plants and shrubs in the understories was estimated by the procedure outlined originally by Pechanec and Pickford (1937).

RESULTS AND DISCUSSION

Pre-Fire

There were no (statistically significant) differences in either soil erosion or deposition on the Cascabel Watersheds throughout the pre-fire period of the study. Therefore, these respective data sets were pooled for analysis of the pre-fire movement of soil on the watersheds.

Soil erosion and deposition measurements in the spring and fall were compared to determine whether seasonal differences in soil movement occurred. It was found that the spring measurements of soil erosion averaging 15.7 tons/acre were (statistically) similar to the 13.4 tons/acre average of the fall measurements. Seasonal depositions of soil differed, however, with the spring measurements averaging of 4.6 tons/acre less than the fall measurements of 7.9 tons/acre. Examination of the data sets indicated that this difference in soil deposition was attributed primarily to the measurements obtained in 2006. Depositions of soil in the spring of 2006 averaged 3.0 tons/acre, while the average of the fall measurements was 9.0 tons/acre. These "large" differences might have been a result of the "large" differences in seasonal rainfall amounts before the measurements were taken. Measurements of soil deposition obtained in the spring of 2006 followed winter rains totaling only 1.6 inches, while those in the fall were obtained after 15.5 inches of summer rainfall. These respective seasonal rainfall totals represented the extremes in the seasonal totals of rainfall for any year in the pre-fire period of measurements.

There were no consistent patterns in the seasonal magnitudes of soil erosion and deposition relative to rainfall amounts, rainfall intensities, rain amounts necessary to generate overland flows of water, or sequencing of the rainfall events. It could be that the values selected from the literature for these "characterizations" of rainfall did not adequately reflect the on-site conditions on the Cascabel Watersheds. Other (unknown) values might have been more appropriate.

There was a significant relationship between soil erosion and deposition and hillslope position. Soil erosion was generally less on the upper slopes, averaging 9.8 tons/acre, than on either the middle or lower slopes averaging 16.0 and 17.1 tons/acre, respectively. This finding could be attributed to the possibility of "thinner soil profiles" on the upper slopes. However, measurements of soil depths on the hillslopes of the Cascabel Watersheds are not available to confirm this possibility. That deposition of soil is often larger on the lower slopes (Rich 1962, Ruhe and Walker 1968) was corroborated by the pre-fire measurements on the watersheds. Seasonal depositions of soil on the lower slopes of the Cascabel Watersheds averaged 8.5 tons/acre in comparison to average depositions of 5.4 tons/acre on the upper and middle slopes combined. Seasonal soil erosion was also greater on slopes in excess of 25 percent, averaging 19.4 tons/acre, than on the lesser slopes averaging of 11.8 tons/acre. However, seasonal depositions of soil on the watersheds were not related to slope steepness. It was unclear why the measured soil movement was greater on northwardly facing aspects than southwardly facing aspects. Parenthetically, there were no significant differences in the densities of trees or production of herbaceous plants on these two general aspects.

Relationships between either seasonal soil erosion or seasonal depositions of soil and the densities of the tree overstories or the production of herbaceous plants and shrubs in the understories were insignificant or had little predictive value.

Post-Fire

Determining the effects of the prescribed burning treatments and wildfire on soil erosion and deposition on the hillslopes of the Cascabel Watersheds was the second part of this study. Comparisons of the post-fire data sets obtained to date showed that there were no significant differences in soil erosion or depositions of soil on the respective watersheds experiencing either the prescribed burning treatments or wildfire. Therefore, the data sets for these respective watersheds were pooled for comparisons with pre-fire values.

There was a difference between the pre- and post-fire estimates of soil erosion on the watersheds that were burned in the cool-season based on the measurements obtained in the spring of 2008, approximately two months following the prescribed burn. Soil erosion on these watersheds averaged 13.4 tons/acre before the burn, while the measurement of soil erosion obtained shortly after the burn was 21.8 tons/acre. However, this initial increase in post-fire soil erosion was short-lived. Measurements of soil erosion following the summer rains of 2008 and those obtained in 2009 were similar to the pre-fire average. A measurement of the deposition of soil on these watersheds in the spring of 2008 was similar to the pre-fire average.

Post-fire soil erosion and deposition in the fall of 2008 and spring and fall of 2009 were similar to the corresponding pre-fire averages in most instances. More specifically, measurements of soil erosion following the two prescribed burning events were within the range of pre-fire values. Absence of widespread water repellent soils after the events (Stropki et al. 2009), and, as a consequence, the likelihood of a little change in the overland flows of water necessary to transport soil particles downslope were the presumed causes for the lack of significant differences in pre- and post-fire soil erosion rates.

Measurements of soil deposition on the watersheds burned by the prescribed treatments were within the range of pre-fire averages. However, there was a difference in the depositions of soil on the watersheds burned by the wildfire in the fall of 2008 relative to the pre-fire average. Soil deposition on these watersheds before the wildfire averaged 5.8 tons/acre, while the deposition of soil measured in the fall of 2008 was 7.6 tons/acre. The reason for this initial difference in soil deposition is not known. Measurements of soil deposition on these watersheds obtained in 2009 were similar to the average of depositions of soil on the watersheds before the wildfire.Soil erosion and deposition following the burning events were not related to the rainfall after these events, physiographic characteristics of the watersheds, or post-burn vegetation on the watersheds.

CONCLUSIONS

The results of this study should be helpful to land managers and other stakeholders interested in re-introducing more natural fire regimes into the oak savannas of the Southwestern Borderlands region. While the three burning events on the Cascabel Watersheds should be considered case studies, a conclusion that the low severity prescribed burning treatments and wildfire impacting the watersheds had relatively little effect to date on soil erosion and deposition in oak savannas in the region appears warranted. It should be remembered, however, that this conclusion is based on measurements of pre- and post-fire soil movement that were made in a period of prolonged drought.

Knowledge of the effects of prescribed burning treatments of other (higher) severities, in other seasons, and on the array of ecosystem resources and factors affecting the hydrologic functioning of oak savannas is necessary before comprehensive, ecosystem-based plans for introducing more natural fire regimes through prescribed burning treatments.

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OCCURRENCE AND PRODUCTION OF AGAVE ON THE CASCABEL WATERSHEDS FOLLOWING THREE BURNING EVENTS

Peter F. Ffolliott,[1] Gerald J. Gottfried,[2] and Steven R. Woods[1]

Agaves are among the more conspicuous plants in the drylands of the Southwestern Borderlands region in almost any landscape whether natural or manipulated. All agaves have succulent or semi-succulent leaves that form rosettes from a few inches to several feet across, but there are many variations to this basic pattern (Turner et al. 1995, Phillips and Comus 2000). Various species of agave are sources of food, fences, rope, medicine, and liquor. Surprisingly, however, little information is available on the effects of disturbances such as the occurrence of fire on the status of agave in the region. This paper reports on the status of *Agave palmeri* following prescribed burning treatments and a wildfire on the Cascabel Watersheds in the Peloncillo Mountains of southwestern New Mexico. Slauson et al. (1999) reported that nectar and pollen from this agave species are the primary foods for the federally listed lesser long-nosed bat (*Leptonycteris curasoae*).

Agave palmeri inhabits rocky slopes in the oak savannas of the Southwestern Borderlands region in southeastern Arizona, southwestern New Mexico and neighboring states in northern Mexico (Irish and Irish 2000). Such a habitat is found on the Cascabel Watersheds. The Rocky Mountain Research Station, U.S. Forest Service and its cooperators established these watersheds to evaluate the impacts of burning events on the ecological and hydrologic characteristics of the oak savannas in the region (Gottfried and Edminster 2005, Gottfried et al. 2007). This paper reports on the effects of cool-season and warm-season prescribed burning treatments and a wildfire on the occurrence and production of *Agave palmeri* on these watersheds.

CASCABEL WATERSHEDS

Twelve small watersheds, ranging from 20 to almost 60 acres in size, in the Peloncillo Mountains of southwestern New Mexico were (collectively) the study areas. The areal aggregation of these watersheds, the Cascabel Watersheds, is 451.3 acres. The watersheds are located between 5,380 and 5,590 feet in elevation (Gottfried et al. 2000). The nearest long-term precipitation station indicates that annual precipitation averages about 23.5 inches, with nearly one-half occurring in the monsoonal summer. However, a prolonged drought was impacting the study areas at the time of this study. Vegetative, geologic, physiologic, and hydrologic characteristics of the Cascabel Watersheds have been described elsewhere (Gottfried et al. 2007) and, therefore, are not presented in this paper.

PRESCRIBED BURNING TREATMENTS AND WILDFIRE

The original objective of the research program on the Cascabel Watersheds was to evaluate the effects of warm-season (May through October) and cool-season (November through April) prescribed burning treatments on the natural resources of these watersheds including understory plants. These evaluations would then be compared to unburned (control) watersheds to determine the effects of these treatments. Four of the watersheds were burned during the cool-season in early March 2008. Three of the four watersheds to be burned in the warm-season were burned on May 20, 2008, with burning of the fourth watershed scheduled for a later date. However, wind gusts up to 60 mph blew firebrands onto the unburned fourth watersheds in the morning of May 21, 2008. The resulting wildfire crossed the boundary lines among the watersheds and then spread beyond the watersheds to burn approximately 4,000 acres.

The original objective of the research program on the Cascabel Watersheds had to be modified, therefore, to accomplish the objective of the program by evaluating the impacts of the prescribed burning treatments and the wildfire on the natural resources. The effects of these three burning events on the occurrence and production of *Agave palmeri* on the watersheds are reported in this paper. Corresponding information was not available before the burning events. This paper, therefore, only reports on the status of this agave species following the burns.

1 School of Natural Resources, University of Arizona, Tucson, Arizona
2 Rocky Mountain Research Station, U.S. Forest Service, Phoenix, Arizona

FIRE SEVERITIES

A system that relates fire severity to the soil-resource response to burning (Hungerford 1996) was the basis for classifying severity of the prescribed burning treatments and the wildfire at the sample plots on the watersheds following the three burning events. Classifications of fire severity at the sample plots were then extrapolated to a watershed-basis to determine the percentages of the watersheds that were unburned or burned at low, moderate, and high fire severities (Stropki et al. 2009). These extrapolations indicated that all of the Cascabel Watersheds had been exposed to a low burning severity regardless of the event.

STUDY PROTOCOLS

On each of the Cascabel Watersheds, between 35 and 45 primary sample plots had been located along transects perpendicular to the main stream systems and situated from ridge to ridge to provide the sampling basis to obtain pre-burning data sets on natural resources (Gottfried et al. 2007). The interval between the plots varied with the size and configuration of the watershed sampled. A total of 421 plots were located originally on the 12 watersheds. This sampling design had been and continues to be used in studies of natural resources on the watersheds.

Occurrence was noted and the production of *Agave palmeri* was estimated on 9.6 ft^2 plots centered over the primary sample plots in the fall of 2009, almost 20 months following the cool-season prescribed burn and 17 months after the warm-season prescribed burn and the wildfire. Information on the occurrence and production of *Agave palmeri* was obtained on 417 of the 421 plots located originally due to missing plots. Production of agave was estimated by the double-sampling method outlined originally by Pechanec and Pickford (1937). Samples of the plant were collected on temporary plots to develop the correction factors necessary to convert field estimates of green weight to actual (oven-dried) weights of production.

RESULTS

Occurrence and production (standing biomass) of *Agave palmeri* on the Cascabel Watersheds following the three burning events were similar. This finding was not surprising, however, because all of the watersheds had been exposed to the same fire severity (Stropki et al. 2009. Therefore, the respective data sets were pooled to describe the occurrence and production of *Agave palmeri* on the watersheds at the time of sampling.

Occurrence

Agave palmeri was found on 54 (12.9 percent) of the 417 plots on the watersheds after the burning events. The occurrence of entire plants or plant parts on these plots was random and non-normal in its distribution on the watersheds. Neither topography (slope position, slope percent, or aspect) nor surface characteristics (percent of rockiness, bare soil, or plant cover) of the plots were related to the occurrence of *Agave palmeri* following the burning events. Presence of the trees, shrubs, or herbaceous plants on the plots was also unrelated to the occurrence of agave.

While none occurred on the plots, an unknown number of burned or partially burned *Agave palmeri* were scattered on the watersheds following the burning events. Similar to the situation observed with the tallied plants," however, there was no apparent pattern in the occurrence of these dead or fire-damaged plants.

Production

Average production (standing biomass) of *Agave palmeri* on the watersheds following the burning events was about 0.25 pounds per acre. This value was a fraction of one percent of the average production of all understory plants (an estimated 134.2 pounds per acre) in the fall of 2009. Average production of the *Agave palmeri* on the individual Cascabel Watersheds ranged from a trace to almost 16.7 pounds per acre. (Average production of *Agave palmeri* on the 54 plots with the plant was 10.7 pounds per acre, ranging from a trace to almost 100 pounds per acre.) The production levels of agave were random and non-normal in their distribution. Neither watershed characteristics nor production of the other plant species on the 9.6 ft^2 plots were related to the production levels of agave.

DISCUSSION

There is a general lack of information on the effects of fire on *Agave palmeri* in the Southwestern Borderlands region. However, in one study, Scott (1999) found an average of 2.39 surviving stalks per acre following the Baker Burn of 1995 in the Peloncillo Mountains. He thought that this post-fire population of agave would be sufficient in providing food to the migrating bats in the region. Slauson (2002) reported later that initial mortality of *Agave palmeri* after a prescribed burn in the same area was

3.3 percent. Slauson indicated that this mortality should not decrease the food reserves for either the migrating bats or reproductive resources and survivorship of the plant. The rosette shape of the plant and an adaption for carbohydrate storage in the center of the rosette protects much the food reserves of the plant from fire. Nevertheless, it has been concluded that a "significant" decline in *Agave palmeri* numbers as a result of a fire could reduce the numbers of bats and other organisms that are dependent on a steady flow of nectar and pollen from the plant (Howell and Roth 1981).

Worthington and Corral (1987) studied the effects of fire on the densities of another agave species following a "cool burn" in the Chihuahuan Desert of western Texas. These investigators tallied the occurrence of *Agave lechuguilla* on "belt transects" approximately 6.5 feet wide and 165 feet long established on burned and unburned sites 16 months following the fire. Five parallel transects were located on each site. Worthington and Corral reported that the density of *Agave lechuguilla* on the unburned site was 24.5 plants per acre, while they tallied no agave on the burned site. They could not conclude explicitly that the fire eliminated the agave on the burned site, however, because occurrence of the plant had not been tallied on either of the sites before the fire.

CONCLUSIONS

A similar "inconclusive" conclusion on the effects of fire on *Agave palmeri* on the Cascabel Watersheds is reached in this study because pre-fire estimates of the occurrence and production of the plant were not available. The results reported in this paper, therefore, represent a "case study" on the status of *Agave palmeri* following the three burning events on the watersheds at the time that the plant was sampled. Further monitoring will be required to determine the long-term effects of these burning events on the plant.

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PATTERNS AND TRENDS IN STREAMSLOW FROM 1939 TO 1980 AT WORKMAN CREEK, SIERRA ANCHA EXPERIMENTAL FOREST, ARIZONA

Peter E. Koestner,[1] Karen A. Koestner,[2] Daniel G. Neary,[2] and Gerald J. Gottfried[1]

Water is a vital natural resource and its paucity in the Southwest makes it of particular importance. There is a long history in the Southwest of research on increasing water yields from forests starting in 1911 at Wagon Wheel Gap Experimental Watershed in central Colorado. Watershed studies in the Salt River Basin began in 1925 with the Summit Plots upstream from Roosevelt Dam. The Summit Plots were established to study the effects of vegetation change and mechanical stabilization on stormflows and sediment yields after it was discovered that 101,000 acre-feet of sediment had accumulated behind Roosevelt Dam in less than two decades (Gottfried et al. 1999). Watershed studies at Workman Creek were initiated in 1938 to investigate the hydrology of mixed conifer forests and determine the effects of manipulating these forests to increase water yields. Streamflow was monitored from 1939 to 1980 at the four gaging stations.

Two watersheds, North Fork, and South Fork, were instrumented with 90° V notch weirs. Main Dam, located downstream of the confluence of the other three watersheds was instrumented with a compound weir consisting of a 90° V-notch weir and a 7-ft Cipolletti weir. The flows of Middle Fork were calculated as the difference between Main Dam and the two other catchments. The objective of the treatments on North Fork was to determine the water yield increases possible from converting a mixed conifer forest cover into a cover of herbaceous plants in stages. The objective of the treatments on South Fork was to determine the hydrologic effect of forest management practices as conducted in the 1950s and 1960s. Middle Fork was the control. The watersheds were calibrated 13 years prior to treatment.

SITE DESCRIPTION

The Workman Creek watershed is located in Central Arizona approximately 30 miles north of Globe in the Sierra Ancha Experimental Forest. The climate at the site is characterized by two wet seasons; one in winter that receives approximately 67 percent of the annual precipitation and the other in summer receives approximately 33 percent of annual precipitation. Precipitation measured on Middle Fork from 1938 through 1973 averaged 32.89 inches annually. Winter precipitation (October-May) averaged 22.01 inches while summer precipitation (June-September) averaged 10.88 inches (Rich and Gottfried 1976). The elevation ranges from 6590 to 7724 feet. The basin drains to the west and is divided into three catchments varying in size from 248 to 521 acres in size, each with a perennial stream (Rich et al. 1961).

Soils are loam to clay loam in texture and granular to crumb in structure. Subsoils are layered and vary in texture from clay loams to clay. Soil depth ranges from a few inches to 15 feet. Soils rest on Dripping Springs quartzite with intrusions of diabase and basalt plugs and sills (Rich et al. 1961).

The pre-treatment forest on the watershed was mixed conifer with ponderosa pine (*Pinus ponderosa*) being the most common tree, especially on the drier sites. Douglas-fir (*Pseudotsuga menziesii*) and white fir (*Abies concolor*) were more common on the wetter sites. The understory shrubs consisted of mostly New Mexican locust (*Robinia neomexicana*) and Gambel oak (*Quercus gambelii*). Riparian trees found sparsely along the stream channel consisted of bigtooth maple (*Acer grandidentatum*), Arizona alder (*Alnus oblongifolia*), Arizona walnut (*Juglans major*), and aspen (*Populus tremuloides*). Table 1 presents proportions of trees by species on the three watersheds. There is a 20-acre meadow on Middle Fork and a 2-acre meadow on South Fork. Other than these two meadows, the herbaceous cover was less than 1 percent of the original ground surface (Rich et al. 1961).

FOREST TREATMENTS AND PREVIOUSLY REPORTED RESULTS

North Fork Watershed Treatments

The treatments on North Fork were designed to determine water yield increases possible in mixed

1 Rocky Mountain Research Station, U.S. Forest Service, Phoenix, Arizona
2 Rocky Mountain Research Station, U.S. Forest Service, Flagstaff, Arizona

Table 1. Proportions of trees by species on the three watersheds within Workman Creek.

Watershed	Total area (acres)	Ponderosa pine	White fir	Douglas fir	Gambel oak	Other	Basal area (ft^2/acre)
		Average tree composition					
North Fork	248	52.3%	20.1%	4.1%	22.2%	1.4%	174
Middle Fork	521	46.4%	20.2%	10.1%	21.8%	1.5%	193
South Fork	318	59.4%	25.5%	5.8%	8.0%	1.3%	201
Total	1087	53.0%	22.0%	7.0%	17.0%	1.0%	191

conifer forests by changing the cover type to grass. The trees on the watershed were removed in a series of steps from the riparian vegetation component, mixed conifer component, and ponderosa pine component, respectively. The first treatment removed riparian trees, mainly Arizona alder and bigtooth maple, from around the stream channels, seeps, and springs. After the trees were cut, the stumps were coated with herbicides to prevent subsequent sprouting, a common regeneration strategy for these species. The riparian cut removed 0.6 percent of the total basal area of North Fork, with no statistically significant increase in annual water yield when compared to the control watershed. However, the treatment did increase summer stormflows and peaks by up 280 percent for rainfall events larger than 2 inches in the 5-year period following treatment (Hibbert and Gottfried 1987).

The second treatment on North Fork converted the moist-site mixed conifer forest, predominantly Douglas-fir and white fir trees, to grass and other herbaceous plants. About 80 acres, approximately 32 percent of the watershed, was clear-cut in 1958. This treatment resulted in water yield increases of 42 percent or 1.26 inches for average conditions (Rich and Gottfried 1976). Increases of 322 to 442 percent were observed for summer stormflows and peaks for storms of 1 to 2 inches. Modest increases in stormflow of approximately 30 percent were reported for winter storms. The percent increases were smaller for winter storms than summer storms but had larger volume increases in water yield (Hibbert and Gottfried 1987).

The third treatment on North Fork converted the 100-acre dry site, predominantly ponderosa pine although Douglas-fir and white fir were also represented, to grass and other herbaceous plants. The timber harvest of about 100 acres was completed in the fall of 1966. A prescribed fire during the winter of 1969 removed much of the remaining stand. The surviving trees were treated with herbicide, completely clearing the stand. This treatment increased water yield 1.32 inches (Rich and Gottfried 1976). Summer stormflows and peaks were similar to the moist site treatment (Hibbert and Gottfried 1987). The combined effect of the treatments was an increase of 2.7 inches or 84 percent in annual streamflow (Rich and Gottfried 1976). The water yield increases remained stable of 13 years (Gottfried et al. 1999).

South Fork Watershed Treatments

The treatments on the South Fork were designed to determine the effects of managing a forested watershed for the production of high-quality timber on water yield and sedimentation. Only the water yield results are discussed in this paper.

The first treatment, a single-tree selection, started in June 1953 and ended in November 1955, reduced the basal area of the forest by 24 percent. The stand-improvement portion of the treatment began in 1956. Small areas of pine infested with mistletoe were treated with herbicide and larger stands were isolated using herbicides to create 60-foot buffers. Gambel oak, New Mexican locust, and firs, which were determined to be competing with pine, were also removed (Rich et al. 1961). These actions reduced basal area a further 6 percent. A wildfire in 1957 burned the 60 acres on the southeast section of the watershed, removing another 9 percent of the total basal area. Other damage associated with logging (i.e., construction of access roads and skid trails) reduced the basal area by 6 percent. All told, basal area was reduced 45 percent. The increases in water yields were statistically significant but of little practical importance (Rich and Gottfried 1976). Summer stormflows and peaks increased "most dramatically" in the two summers following the wildfire. Stormflow volumes increase 2.5 to 3 times, while stormflow peaks increased 5 to 10 times. However, after two summers, stormflow volumes and peaks returned to near pretreatment levels (Hibbert and Gottfried 1987).

The second treatment on South Fork took place in 1966. Before implementation of this treatment, basal area on the watershed was 118 ft^2 /acre. Merchantable timber was removed and the remaining trees were thinned to a density of 40 ft^2 /acre. Areas infested with dwarf mistletoe and areas of fir were cleared, windrowed, and burned. About 55 acres of thinned pine remained following this treatment, with the rest of the watershed mostly cleared. Reforestation efforts were largely frustrated by plant competition and activities of pocket gophers (Thomomys bottae). This treatment significantly increased water yield (Rich and Gottfried 1976). Summer stormflows and peaks were less responsive than the selection cut treatment, except for storms of less than 1.5 inches in which case the responses were about equal. A minor increase in winter stormflows and peaks were attributed to the treatment (Hibbert and Gottfried 1987).

The combined these treatments produced an increase in water yield of 111 percent or 3.67 inches that was sustained for 13 years (Gottfried et al. 1999, Rich and Gottfried 1976). Table 2 summarizes all of the treatments.

The weirs were de-commissioned in 1980 due to the U.S. Forest Service shifting their priorities (Gottfried et al. 1999). However, the weirs were reactivated after the Coon Creek Fire of 2000, a 9,644-acre fire, burned the three watersheds (Gottfried et al. 2003). The patterns and trends in streamflow at the Main Dam from 1939 to 1980 are described in this paper.

METHODS

Workman Creek streamflow data-sets were obtained from archived files at the Rocky Mountain Research Station Flagstaff Laboratory. Each water year was a complete file of daily and monthly summaries. To make the Sierra Experimental Forest legacy data available (via the Rocky Mountain Research Stations website1), the annual summaries of monthly streamflow between 1939 and 1980 were digitized into Excel for ease of analysis and integration.

After digitization, the yearly, seasonal (winter and summer), and monthly means of streamflow were calculated for a given water year. A water year (WY) spans October 1 through September 30 of the following year and is named for the year in which it ends, that is, WY 1939 is from October 1, 1938, through September 30, 1939. Winter flows occur between October 1 and May 31, while summer flows between June 1 and September 30 for a given year.

Precipitation data were obtained from the Western Regional Climate Center. These data were collected at a site that is approximately 3 miles from Main Dam and about 1400 feet lower in elevation. Though there is a discrepancy in elevation, this was the closest gauging station with long-term records for use in determining trends in streamflow at Workman Creek.

RESULTS AND DISSCUSSION

Average yearly runoff at Workman Creek from 1939 to 1980 was 4.53 inches. The lowest annual runoff of 0.94 inches was recorded in 1955, while the highest annual runoff of 18.36 inches was recorded in 1980 (Figure 1).

Table 2. Treatment types and application dates on the North Fork and South Fork watersheds of Workman Creek (Rich and Gottfried 1976). Middle Fork was the untreated control.

	Treatment	Year	Area treated (acres)	
North Fork	Riparian Cut	1953	1.50	0.6%
	Moist Site Cut	1958	79.36	32%
	Dry Site Cut	1966	99.2	40%
	Treatment	**Year**	**Basal area removed (ft^2/acre)**	
South Fork	Single Tree Selection	1953-1955	72.38	36%
	Wildfire	1957	18.09	9%
	Thinning treatment	1966	70.35	35%

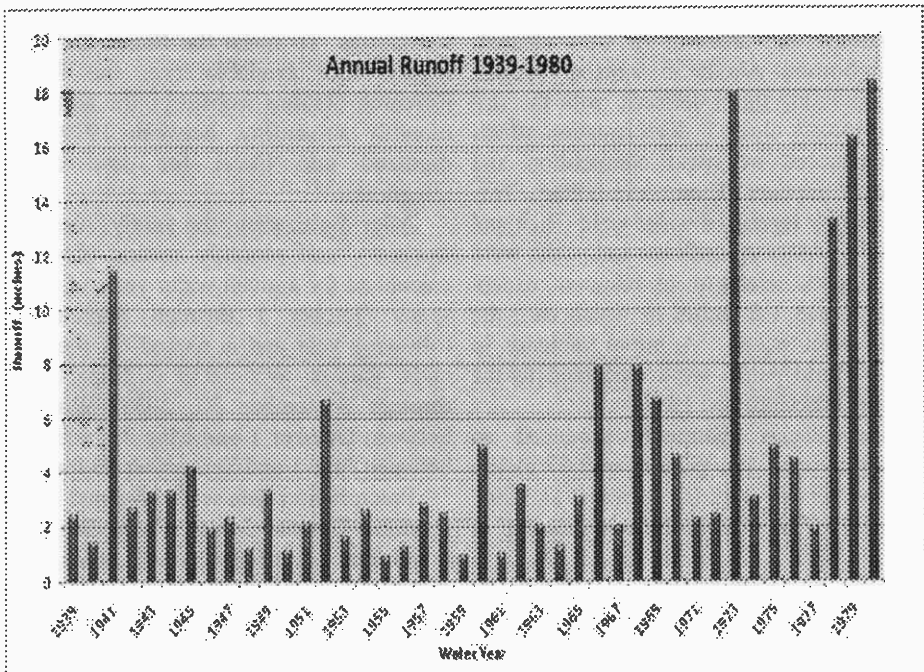

Figure 1. Total yearly runoff at Main Dam of Workman Creek from 1939-1980.

A comparison of average monthly precipitation and runoff at Workman Creek is shown in Figure 2. The general bimodal precipitation pattern of the region and the occurrence of the peak runoff volumes throughout the winter and early spring months on Workman Creek are illustrated by this figure.

Due to the bimodal precipitation pattern of this region, the data were lumped into winter runoff and summer runoff. Winter runoff accounted for 89 percent of total runoff and winter precipitation for 66 percent of total yearly precipitation. Summer runoff was 11 percent of total yearly runoff and summer precipitation was 33 percent of annual precipitation. These data sets indicate that winter precipitation was the "primary driver" of runoff. The relationship between total runoff and total precipitation has an R^2 value of 0.74, while the relationship between total runoff and winter precipitation has an R^2 value of 0.84. The relationship between total runoff and summer precipitation has an R^2 value of 0.15.

The fact that winter precipitation has the "strongest relationship" to total yearly runoff is due likely to the high amount of winter precipitation and less water lost to evapotranspiration (ET). Summer runoff is controlled by the high ET demands during those months and the consequent high soil-recharge demands. During the summer there is a "weak correlation" between runoff and precipitation (R^2 = 0.03). In the winter months, the coefficient of determination between precipitation and runoff was highly significant (R^2 = 0.86). This same pattern has been shown in other mixed conifer forests. For example, Franco-Vizcaino et al. (2002) analysis of worldwide streamflow and ET values showed that as watershed-systems become more water stressed, such as the case in summer months, the positive relationship between ET and precipitation increases while the correlation of precipitation and streamflow decreases.

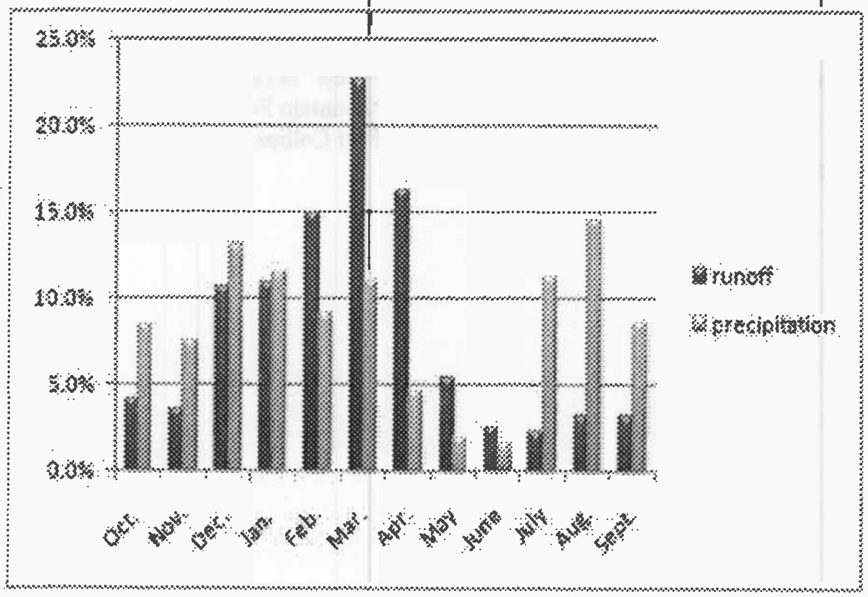

Figure 2. Comparison of average monthly precipitation and runoff at Workman Creek. Data are presented as monthly percents of total of the annual precipitation and runoff to illustrate the trend therein. Due to the location of the precipitation gauge at a lower elevation, the actual quantities of precipitation are not included. Runoff in winter months is related more to snowmelt than to precipitation received during that month.

CONCLUSIONS

This analysis of Workman Creek was made possible because of the many years that Rocky Mountain Research Station personnel spent working at Workman Creek, and, therefore, it underscores the necessity for long-term research projects. To understand long-term trends, a commitment must be made to long-term research. Without long-term records, information of the possible effects of climate change is impossible. The 20-year gap in the records at Workman Creek was unfortunate. However, a "representative record" for this watershed is being obtained through the data-analysis presented in this paper and the data currently being collected on the watershed. Not only are streamflow-measurements continuing, but air-pollution monitoring equipment is being installed on the Sierra Ancha Experimental Forest to further increase our knowledge of the climate of this mid- elevation watershed.

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A COMPARISON OF RIPRIAN DATA COLLECTED ON THE CORONADO NATIONAL FOREST 1998-2009

Robert E. Lefevre[1]

Riparian areas on the Coronado National Forest have been monitored periodically since 1986. U.S. Forest Service personnel have carried out this monitoring effort as part of the 1986 Forest Plan (USDA 1986). As part of the monitoring, substrate particle size and width to depth ratio were measured. Changes in these measurements between 1998 and 2009 were observed. Possible causes for these changes are described in this paper.

Table 1: Watershed properties for riparian monitoring locations.

Watershed	Total Acres	Percent Burned	Water Slope (percent)	Geology	Uses	History
Canada del Oro	14,496	100%	14%	Granitic, alluvium	Recreation, Grazing	Burned 2003
Cave Creek	25,547	46%	7%	Rhyolite	Recreation, Grazing	Burned 1994
Lower Lyle Canyon	14,022	1%	4%	Granitic, alluvium	Recreation, Grazing	Burned 2009
Lower Redrock Canyon	17,955	29%	2%	Rhyolite, alluvium	Recreation, Grazing	Burned 2002
Lower Scotia Canyon	1,630	35%	7%	Granitic	Recreation, Grazing	Burned 1994
Middle Redrock Canyon	13,836	30%	3%	Rhyolite, alluvium	Recreation, Grazing	Burned 2002
Middle Rucker Canyon	8,683	51%	8%	Rhyolite	Recreation, Grazing	Burned 1994
North Fork Cave Creek	1,180	100%	24%	Rhyolite	Recreation, Grazing	Burned 1994
Sabino Canyon	21,011	89%	8%	Granitic	Recreation	Burned 2003
Upper Lyle Canyon	1,402	8%	8%	Granitic	Recreation, Grazing	Burned 2002
Upper Redrock Canyon	7,407	26%	2%	Rhyolite, alluvium	Recreation, Grazing	Burned 2002
Upper Rucker Canyon	4,667	66%	8%	Rhyolite	Recreation, Grazing	Burned 1994
Upper Scotia Canyon	1,108	4%	7%	Granitic	Recreation, Grazing	Burned 2002

METHODS

Study Area

Over 360 data collection points have been monitored in channels of the Coronado National Forest. Nearly all are ephemeral or intermittent in nature. Thirteen generally intermittent channels were chosen for this comparison. These were chosen because the Riparian Area Survey and Evaluation System (RASES) was the data collection method (USDA 1989) used for all sites. At first it was also believed that four of these sites represented channels of unburned watersheds. However, it was later determined that all sites in had at least partially burned since 1994. Further examination of the entire set of data exposes the fact nearly every watershed on the Coronado National Forest has experienced fire to some extent since 1994. Table 1 presents some data about each of the thirteen sample points used in this comparison.

Channel Substrate

Pebble counts conducted for the RASES method separated particle sizes into 7 size classes: silt/clay (<0.62 mm), fine sand (0.62 mm-1 mm), coarse sand (1.01 mm-2 mm), gravel (2.01 mm-64 mm), cobble

1 Coronado National Forest, U.S. Forest Service, Tucson, Arizona

(64.01 mm-256 mm), boulder (256.01 mm-2048 mm), and bedrock (>2048 mm). The statistic used to discuss substrate is D50, which is the substrate particle size (mm) at which 50% of the sample is finer.

Width to Depth Ratio

Channel measurements were used to determine the width to depth ratio. Channel width was measured at the apparent bankfull elevation. Channel depth is the mean depth as derived from cross section measurements.

Watershed Size

The watersheds contributing to these data vary from 1,108 to 21,011 acres in size. This variable (alone) prohibits meaningful comparison of data sets between points. Watershed size contributes to channel morphology attributes by providing more or less water to the site. Other variables including drainage pattern, geology, watershed uses, and watershed slope also contribute to channel morphology. In an effort to keep these variables at a minimum, the watersheds chosen for comparison have generally dendritic drainage patterns, are located on granitic or rhyolite formations, have recreation and cattle grazing as primary uses, and (generally) have a slope between 4% and 8%. Two watersheds, Canada del Oro and North Fork Cave Creek, have steeper slopes.

History

The watersheds of all the riparian areas sampled have been at least partially burned since 1994. Observations made by looking at the data are that channels of more recently burned watersheds exhibit substrate and width to depth ratios that appear to be unstable compared to those that burned longer ago. Data collected on watersheds that burned since 1998 were collected before and after the burn event. Data collected in watersheds that burned in prior to 1998 were (of course) collected since the watershed was burned.

RESULTS

Channel Substrate

All data collection locations exhibit measured changes in the D50 channel substrate measurements. Of the 13 locations, 8 have D50 in the same particle size class (gravel for example) for both sample dates in spite of the difference in measured sizes. Two locations had a larger size class in 2009 than in 1998. Table 2 presents the particle size measurements for the respective sample date.

Width to Depth Ratio

As with the substrate data, all data collections exhibited measured changes in width to depth ratio. Of the 13 locations, 5 were similar on both sample dates, 4 were smaller, and 4 were larger. The width to depth ratios are shown in Table 3.

DISCUSSION

Channel Substrate

Maintaining the same particle size class can be interpreted as an indicator of a stable watershed and channel. Of those channels that retained the same particle size class over both sampling dates, all except Lower Lyle Canyon have been recovering from fire within the watershed for more than 8 years. Lower Lyle Canyon was burned in 2009, but only to the extent of 1% of the watershed upstream from the sample location. Of the three channels that have had a change in size class from a larger size class to a smaller one, two locations were in watersheds that burned severely in 2003. These two, Sabino Canyon and Lower Canada del Oro, appear to have active sediment transport occurring during recent runoff events. Lower Redrock Canyon had a measured change in D50 from 4.5 mm to 1 mm, technically a change from the finest gravel designation to coarse sand. There is no evidence of degrading conditions such as pools filling with fine material as there is Sabino Canyon and Lower Canada del Oro in Lower Redrock Canyon. The two locations that have larger particle size classes were burned in 1994. There is evidence that these locations had elevated levels of fine material move through the system after fire, but watershed recovery has provided for fine material to be moved out, leaving coarser material in recent events.

Width to Depth Ratio

Width to depth ratio can be used to interpret channel stability. Large width to depth ratios indicate more erosive energy stress is being placed near the banks compared to smaller width to depth ratios (Rosgen and Silvey 1994). Of those channels that maintained a similar width to depth ratio for both sample dates, watershed history is variable, ranging from burning in 1994 to burning in 2003. Those channels developing larger width to depth ratios also have a variable history, but all were burned since 2002 indicating a generally shorter recovery period. The channels with developing smaller width to depth ratios were all burned prior to 2003, indicating a generally longer recovery period.

Table 2: Particle size measurements.

Channel	D50 1998 (mm)	D50 2009 (mm)	Particle Size Class
Canada del Oro	105	1	Changed from Cobble to Sand
Cave Creek	28	34	Gravel
Lower Lyle Canyon	8	12	Gravel
Lower Redrock Canyon	4.5	1	Changed from Gravel to Sand
Lower Scotia Canyon	30	10	Gravel
Middle Redrock Canyon	7	2	Gravel
Middle Rucker Canyon	20	64	Changed from Gravel to Cobble
North Fork Cave Creek	23	500	Changed from Gravel to Boulder
Sabino Canyon	90	16	Changed from Cobble to Sand
Upper Lyle Canyon	7	45	Gravel
Upper Redrock Canyon	7	10	Gravel
Upper Rucker Canyon	22	32	Gravel
Upper Scotia Canyon	0.1	1	Sand

Table 3: Width to depth ratios.

Riparian Area	width:depth ratio 1998	width:depth ratio 2009
Canada del Oro	46.9	48.5
Cave Creek	33.4	24.2
Lower Lyle Canyon	65.6	89.2
Lower Redrock Canyon	26.5	21.1
Lower Scotia Canyon	18.5	17
Middle Redrock Canyon	30.3	40.7
Middle Rucker Canyon	20.2	24.5
North Fork Cave Creek	13.5	6.5
Sabino Canyon	66.5	105.4
Upper Lyle Canyon	14.1	8.2
Upper Redrock Canyon	33.5	43.9
Upper Rucker Canyon	34.1	30.6
Upper Scotia Canyon	15.7	8.2

CONCLUSIONS

There are many variables to consider when assessing channel stability following a wildfire in the watershed. The size of the watershed, watershed slope, and extent of the wildfire all play a role in channel stability. Time, however, appears to be a recurring factor for both substrate particle size and width to depth ratio recovery. Longer time periods since a disturbance generally produce data indicating more stable conditions.

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HIGH-ELEVATION WET MEADOWS IN ARIZONA: DISTRUBUTION, THREATS, AND BIODIVERSITY

James A. Allen[1] and Karissa M. Ramstead[1]

High-elevation riparian meadows occur in numerous locations throughout northern, eastern, and southeastern Arizona. They are often referred to simply as wet meadows. For the purposes of this paper, we define "high-elevation" to be >2000 m and therefore differentiate wet meadows from cienegas and other wetlands found at lower elevations (cf. Hendrickson and Minckley 1984).

Where they have not been excessively altered, sedges (*Carex spp.*), rushes (*Juncus spp.*) and other wetland-obligate plant species tend to be dominant in Arizona's wet meadows (Patton and Judd 1970, Judd 1972, Medina 1996). Willow (*Salix*), alder (*Alnus*) and other woody species may also occur in these meadows. Wet meadows typically experience shallow flooding and/or high water tables throughout much of the spring and early summer (Allen-Diaz 1991, Castelli et al. 2000).

The assemblage of vegetation that is found in wet meadows, along with ready access to water, provides good habitat for a wide variety of wildlife species (Hoffmeister 1956, Mecham 1968, Patton and Judd 1970). It is also likely that wet meadows perform many of the same ecosystem and hydrologic functions associated with other wetland types, such as water quality improvement, reduction of flood peaks, and carbon sequestration (Mitsch and Gosselink 2007). Their perceived hydrologic functions, particularly the storage and gradual release of water, are one of the key factors driving the efforts to restore wet meadows currently underway in the Sierra Nevada Mountains of California (National Fish and Wildlife Foundation 2009).

We briefly review the distribution of wet meadows in Arizona and describe some of the major threats to these ecosystems in this paper. The main focus, however, is on the biodiversity values of wet meadows and, in particular, species that occur in wet meadows which have some level of federal or state protection.

DISTRIBUTION

While many individual meadows exist in the state, their total area is relatively small. Less than 1 percent of the landscape in the state is characterized as wetland (Dahl 1990) and wet meadows are just one of several wetland types that occur. The only estimate of the extent of wet meadows that has been located to date is by Patton and Judd (1970), who reported that approximately 17,700 ha of wet meadows occur on national forests in Arizona and New Mexico. While they are believed to be most extensive in the White Mountains, wet meadows are also common throughout the ponderosa pine and mixed conifer forest belt that extends to the northwest of the White Mountains and some also occur within the "Sky Islands" in the southeastern part of the state (Hoffmeister 1956, Ricketson 1990).

THREATS

Despite their apparent value, wet meadows appear to be one of the most heavily altered types of ecosystems in Arizona. Among other things, they have been used extensively for grazing livestock, have become the site of many small dams and stock tanks, have had roads built through them, and have experienced other types of hydrologic alterations, most notably the lowering of their water tables due to stream downcutting, surface water diversions, or groundwater withdrawal (Allen-Diaz 1991, Zeedyk 1996, Long 2002).

Some wet meadows appear to have been affected by the regional-scale incisional event at the end of the 19th century (Cooke and Reeves 1976, Anderson et al. 2003). This event may have been caused by a combination of drought and land use change, most notably heavy cattle grazing, followed by a period of wet conditions (Graf 1988, Hereford 2002). Wet meadows have been impacted, and in some cases possibly created, by large-scale fires in adjacent uplands, followed by the deposition of sediments and woody material on valley floors (Joyal 2004, Long and Burnette 2004). It is also possible that the extirpation of beaver in the Southwest led to profound changes in wet meadow environments by reducing flooding and potentially facilitating some of the stream downcutting that has occurred in so many riparian areas (Parker et al. 1985).

1 Northern Arizona University, Flagstaff, Arizona

Impacts resulting from the types of disturbances described above may include severe erosion, soil compaction, greatly altered hydrology, shrub and tree encroachment, and other changes in plant species composition and diversity (Martin 1999, Medina and Long 2004). Many of the wet meadows that the authors have investigated personally in northern Arizona are in such a degraded state that they no longer appear to be functional wetlands – they flood less regularly and have deeper water tables, soil conditions are no longer frequently anaerobic, and the plant species composition has shifted from wetland obligates to species such as Kentucky bluegrass (*Poa pratensis*).

BIODIVERSITY

We searched the Arizona Natural Heritage Program's Heritage Data Management System for species that occur in wet meadows and that have some level of state or federal protection status. The search included all taxonomic groups in the database with the exception of invertebrates. Species treated below are those that are federally listed as Endangered or Threatened, are on the Forest Service Sensitive Species list for Region 3, are protected by the Arizona Native Plant Law, or are listed as Wildlife of Special Concern (WSC) in Arizona. Because of space limitations, some statements are made below about a species and no citation is included; the source of the information in such cases is the appropriate species abstract available on the Arizona Natural Heritage Program website (http://www.azgfd.gov/w_c/edits/species_concern.shtml, accessed in May 2010). This should be considered as a preliminary effort to catalog the species of concern that occur in wet meadows. Over time we expect that additional species will be identified that should be included; it is also possible that some of the species we have identified do not have a strong enough association with wet meadows to justify further consideration in this context.

Plant Species

Thirteen plant species found in wet meadows have some type of protection status, with most being either on the Forest Service Sensitive Species list and/or having salvage-restricted status in Arizona (Table 1). They are distributed in all parts of the state where wet meadows are found and three are endemic to Arizona. Some of these species (e.g., *Salix arizonica* and *S. bebbiana*) have been the subject of considerable research attention and restoration efforts (e.g., Prendusi et al. 1996).

Animal Species

Fish

Natural fish species assemblages in high-elevation streams, including those that course through wet meadows, apparently were never highly diverse. For example, above 2,000m in the White Mountains, some streams had only Apache Trout (*Oncorhynchus apache*) and speckled dace (*Rhinicthys osculus*) (Clarkson and Wilson 1995). Several other species, such as desert suckers (*Catostomus clarki*) and bluehead suckers (*C. discobulus*), intermingled with these two species at lower elevations (up to or slightly exceeding 2000 m).

Four fish species known to occur in high-elevation riparian settings, the Apache Trout, Gila Trout (*O. gilae*), Little Colorado spinedace (*Lepidomeda vittata*) and Loach minnow (*Tiaroga cobitis*), are federally listed as Threatened. These species are threatened both by habitat loss and alteration, as well as the introduction of several species of non-native trout.

Amphibians and Reptiles

One species known to use wet meadows, the Chiricahua leopard frog (*Rana chiricahuensis*, Mecham 1968), is federally listed as Threatened. The northern leopard frog (*R. pipiens*) is a WSC species. While neither of these species is totally reliant on wet meadows, they may be locally important as breeding, migration or overwintering sites.

At least three other aquatic species of concern in Arizona have been reported to occur above 2,000 m. They conceivably could occur in wet meadows, although their primary habitat is at lower elevations and/or in other habitat types. These include the Sonora tiger salamander (*Ambystoma tigrinum stebbensi*, Endangered), the Northern Mexican garter snake (*Thamnophis eques megalops*, WSC) and the narrow-headed garter snake (*T. rufipunctatus*, WSC).

Birds

Although many bird species are known to use wet meadows, few species of special concern in Arizona appear to be reliant on them. While most often found at lower elevations, the Southwestern willow flycatcher (*Empidonax traillii extimus*, Endangered) can be found up to at least 2600 m in Arizona; Sogge et al. (1997) describe one of four general habitat types in which the species is found as dense stands of willow interspersed with standing water and wet meadow vegetation. Another species of concern in Arizona is the bobolink (*Dolichonyx oryzivorus*, WSC), which has been known to breed in a limited number of wet meadows and other sites in Apache and Gila Counties (Corman and Wise-Gervais 2005). The isolated breeding sites in Arizona are hundreds of kilometers south of any other known breeding populations of this species (Martin and Gavin 1995).

No other bird species of special concern are known to breed in wet meadows, but several others may make use of them as food sources. These include the Mexican spotted owl (*Strix occidentalis lucida*, Threatened), which may occasionally feed on voles and other small mammals found in wet meadows, and the northern goshawk (*Accipter gentilis*, WSC), whose home range has been shown to frequently include wet meadows in other states (Graham et al. 1999, Daw and DeStefano 2001).

Table 1: Protected plant species found in Arizona's high-elevation wet meadows.

Plant Species	Protection Status	Elevation (m)	Range in Arizona	Total Range
Allium plummerae (Plummer Onion)	State Salvage Restricted	1465 - 2745	Four locations in southeastern Arizona	Southeastern Arizona and adjacent northern Mexico
Carex chihuahuensis (Chihuahuan sedge)	Forest Service Sensitive	1100 - 2195	Southeastern Arizona	Southeastern Arizona, Hildago Co., New Mexico, Sonora and Chihuahua, Mexico
Castilleja mogollonica (Mogollon Paintbrush)	Forest Service Sensitive State Salvage Restricted	2590 - 2895	White Mountains, Mount Baldy vicinity, Apache Co.	Arizona endemic
Cypripedium parviflorum var. pubescens (Yellow Lady's Slipper)	State Highly Safeguarded	1830 - 2915	Apache, Graham and Greenlee Co.	From Newfoundland to British Columbia, south to Georgia, Arizona, and Washington, excluding OR, CA, NV, FL, LA, and OK. Also in Europe.
Helenium arizonicum (Arizona Sneezeweed)	Forest Service Sensitive	1830 – 2440	Mainly in Coconino Co. but also in Apache, Gila and Navajo Co.	Arizona endemic
Lilium parryi (Lemon Lilly)	State Salvage Restricted	1675 - 2380	Chochise Co.	Arizona, San Gabriel and San Bernardino Mountains of southern California, and Sonora, Mexico
Pilates gentryi (Mexican bare-ray-aster)	Forest Service Sensitive	1800 – 2800	Huachuca Mountains	Southern Arizona and central Mexico to Chihuahua in Mexico's Sierra Madre Occidental
Rumex orthoneurus (Blumer's Dock)	State Highly Safeguarded Forest Service Sensitive	1365 – 2945	East-central to Southeastern Arizona	East-central to Southeastern Arizona and the Gila, Baldy and Pecos Wilderness areas in New Mexico
Salix arizonica (Arizona Willow)	State Highly Safeguarded Forest Service Sensitive	2605 - 3050	White Mountains, Apache Co.	Apache County, parts of New Mexico and southern Utah.
Salix bebbiana (Bebb's Willow)	Maryland-Endangered	2440 - 3355	Coconino, Apache, and Navajo Co.	Canada and the United States excluding southeastern United States
Trichostema micranthum (Small-flower bluecurls)	State Salvage Restricted	1800 - 2300	Mount Trumbull, northern Mohave Co.	Baja California, southern California, western Arizona
Trifolium neurophyllum (Mogollon Clover)	Forest Service Sensitive	1980 - 2745	White Mountains of eastern Arizona	White Mountains of Eastern Arizona and Mogollon Mountains of New Mexico
Triteleia lemmoniae (Mazatzal Triteleia)	State Salvage Restricted	975 - 2340	Apache and Navajo Co., also parts of Coconino, Gila and Yavapai Co.	Arizona endemic

Mammals

Several small mammal species of special concern breed in or otherwise rely on wet meadows. The Arizona montane vole (*Microtus montanus arizonensis*) is one species closely associated with wet meadows; it is listed as Endangered in New Mexico and is on the Forest Service's Sensitive Species list, but currently is not listed as a species of special concern in Arizona. The New Mexican meadow jumping mouse (*Zapus hudsonius luteus*, WSC) does much of its foraging for food in wet meadow vegetation. The American water shrew (*Sorex palustris*, WSC) occurs in wet meadows and other riparian habitats in the White Mountains -- more than 320 km from the nearest populations of the species in Utah and New Mexico.

CONCLUSIONS

Virtually all of the species mentioned above have reached the stage where they need (or may soon need) special protection because of the alteration of their habitat, either through direct physical manipulation or through indirect means such as the introduction of non-native species. Ultimately, we believe that the single most important threat to wet meadows and most of the species that depend on them is altered hydrology; typically this has meant a shift towards reduced flooding and lower water table levels, to the point where many former wet meadows are no longer technically wetlands. A concerted effort to restore their hydrology seems like the most critical step towards the restoration of functional wet meadows. However, a systematic effort to inventory and assess the condition of Arizona's high-elevation wet meadows might be needed first to better define the scope of the problem and to help justify an increased investment in their restoration.

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RODEO-CHEDISKI WILDFIRE: A SUMMARY OF IMPACTS

Peter F. Ffolliott[1], Cody L. Stropki[1], Hui Chen[1], and Daniel G. Neary[2]

The 462,600 acre Rodeo-Chediski Wildfire of 2002 was the largest known in Arizona's history and the ninth largest wildfire in the United States (National Interagency Fire Center 2009) in terms of the 462,600 acres impacted by the burn.. It damaged or destroyed ecosystem resources, disrupted hydrologic functioning, and altered loadings of flammable fuels in much of the ponderosa pine (*Pinus ponderosa*) forests exposed to the burn. An opportunity to study the impacts of this wildfire on ecosystem resources, hydrologic functioning, and flammable fuels presented itself on the Stermer Ridge Watersheds near Heber, Arizona. These watersheds were instrumented as a cooperative endeavor of the School of Natural Resources and the Environment, University of Arizona, Tucson, and the Rocky Mountain Research Station, U.S. Forest Service, to obtain baseline information on watersheds located on sedimentary soils (Ffolliott and Baker 1977). This information complemented the results obtained from "experimental watersheds" located in ponderosa pine forests on basaltic soils. One of the Stermer Ridge Watersheds was exposed to a high severity (stand-replacing) fire and the other to a low severity (stand-modifying) fire (Ffolliott et al. 2008, 2009). This five-year study (2002 through 2007) was initiated shortly after cessation of the wildfire and continued through implementation of salvage cutting and fuel reduction treatments on the watershed burned by a high severity fire.

A more comprehensive report of the impacts of this historical wildfire on ecosystem resources, hydrologic functioning, and loadings of flammable fuels has been prepared (Ffolliott et al. 2010). The purpose of this paper is to present a brief summary of these impacts.

IMPACTS ON ECOSYSTEM RESOURCES

The Rodeo-Chediski Wildfire impacted on the structures of tree overstories on the Stermer Ridge Watersheds, with the changes more drastic on the watershed experiencing a high severity fire than on the watershed burned by a low severity fire (Ffolliott et al. 2008). The pole class of trees (8 to 10 inches dbh) suffered higher mortality than trees in the sawtimber classes (12 inches dbh and larger). Most of the trees surviving the wildfire but severely damaged by the burn died within two years of the burn and many had started falling to the ground at this time. Salvage cutting and fuel reduction treatments imposed on the watershed burned by a high severity fire eliminated most of the remaining standing trees on this watershed. Stocking of ponderosa pine seedlings after the wildfire was insufficient to sustain a ponderosa pine forest on either of the watersheds.

Production of herbaceous plants and the grass and forb components of herbaceous plants were greater on the watershed burned by a high severity fire throughout the study. An earlier study in southwestern ponderosa pine forests found that production of herbaceous plants generally increases when densities of competing tree overstories decrease (Bojorquez-Tapia et al. 1990). Burning intensities that consume the litter and large amounts of the duff layer can also increase the level of herbage production in these forests (Clary et al 1968). Such a fire occurred on the severely burned watershed.

Mullein ((*Verbascum thapsus*) represented almost one-half of the production of the forbs on the severely burned watershed at its peak occurrence, while this early-seral species contributed less to the lower level of forb production on the watershed burned by a low severity fire. Mullein then declined in its occurrence after dominating the forb component on both of the watersheds in the early and middle years after the wildfire.

Impacts of the Rodeo-Chediski Wildfire on indigenous ungulates and small mammals inhabiting the Stermer Ridge Watersheds at the time of the wildfire differed in magnitude and duration. The increased presence of elk on the watershed burned by a high severity fire in the third year after the burn was attributed to the increase in preferred forage plants (Kufeld 1973, Thill et al. 1983), with availability of a cover being a secondary factor of importance. The smaller quantities of browse preferred by mule deer (Severson 1981, Thill et al. 1983) resulted in lower occurrences of this ungulate relative to that of elk on

1 University of Arizona, Tucson, Arizona
2 Rocky Mountain Research Station, US Forest Service, Flagstaff, Arizona

both of the watersheds. The limited presence of cottontail on both of the watersheds was a result of the limited availability of preferred forage plants (USDA Natural Resources Conservation Service 1999) available to this mammal. Extirpation of Abert's squirrel from the watershed burned by a high severity fire was caused largely by the loss of preferred feed trees (Ffolliott and Patton 1978). Following their initial disappearance, feeding activities by the squirrel were observed on the watershed burned by a low severity fire two years after the wildfire. This finding was attributed largely to most of their preferred feeding trees surviving the fire.

Not knowing the structure of bird communities on the watersheds before the wildfire precluded comparisons of pre- and post-fire species and numbers. Nevertheless, the numbers of birds in some guilds declined, while birds in other guilds were affected less by the wildfire (Ffolliott et al. 2009). Because of the varying responses of birds to the Rodeo-Chediski Wildfire, a mosaic of habitats on a scale much larger than that of the Stermer Ridge Watersheds seems necessary to sustain a "full suite" of bird species following the burn.

IMPACTS ON HYDROLOGIC FUNCTIONING

One impact of the wildfire on the hydrologic functioning of the Stermer Ridge Watershed burned by a high severity fire, and the limited areas of the watershed exposed to a low severity fire but burned at a high severity, was a decrease in the interception of precipitation with the loss of tree overstories and the litter and duff layers by the burn. Reforestation on these severely burned areas in the near future is unlikely without artificial regeneration. Litter and duff layers cannot re-form on the sites experiencing a high severity fire in the near future because of the absence of trees.

Transpiration is another component of the water budget that was altered by the wildfire. To estimate of the magnitude of this hydrologic component, a set of measurements of transpiration rates were obtained by the sap flow method (Swanson 1994). Three severely damaged and three undamaged ponderosa pine trees 18 to 24 inches in dbh were measured on each of the watersheds shortly after the wildfire. While sap flow was detected in all of the trees, the three repeated measurements of each tree sampled were insufficient to reliably differentiate the magnitude of transpiration losses on the two watersheds with confidence. Nevertheless, it was concluded that the loss of water by transpiration was reduced (to unknown levels) on both of the Stermer Ridge Watersheds, with comparatively higher losses on the watershed experiencing a high severity fire because of the greater loss of trees.

Infiltration of water into the soil was not measured in the study. It has been assumed, however, that the formation of water repellent soils as a result of the wildfire inhibited the infiltration rates, with much of the precipitation reaching the soil surface increasing the flows of overland water and concurrent erosion and deposition of soil particles (Garcia-Chevesich et al. 2004). This phenomenon was more pronounced on the watershed burned with a high severity fire because of the more extensive formation of strong water repellency (Stropki et al. 2005). Water repellent soils persisted up to three years after the wildfire, after which it was likely that infiltration of water into the soil increased toward pre-fire levels, while the accelerated overland flows of water and soil movement declined.

Destruction of the vegetative covers and litter and duff layers coupled with formation of the water repellent soils on the watersheds combined to result in peak stormflows of historical magnitudes following the wildfire (Ffolliott and Neary 2003). The highest of these peak stormflows were observed following high intensity, short-duration rainfall events shortly after cessation of the wildfire. While post-fire stormflows of "exceedingly high peaks" are often characteristic of flooding regimes, downstream flooding following the Rodeo-Chediski Wildfire did not occur. Quality characteristics of the water flowing from the watersheds were not measured in this study. It was (ultimately) determined, however, that the concentrations of sediment, nutrients, and other debris in water flowing from the larger area burned by the wildfire presented few environmental problems.

IMPACTS ON FLAMMABLE FUELS

Measurements made one year following the wildfire indicated that 40 percent of the accumulated pre-fire flammable fuels on the watershed burned by the high severity fire had been consumed in the burn, with most of this loss in standing trees. Nearly 75 percent of the pre-fire trees had died by 2005, three years after the wildfire, with this mortality translating into an additional increase of 15 percent in loadings of standing dead trees in relation to that one year after the fire. Measurements of flammable fuels on the watershed exposed to a low severity fire one year after the wildfire indicated that about 20 percent of the pre-fire loadings of flammable fuels had been consumed by the wildfire. The fire killed less than 5 percent of the pre-fire trees on this watershed. This finding was not surprising, however, as much of this watershed that had been burned was impacted by surface fire. The greater loadings of litter and duff on this watershed at this time was attributed to the litter falling from standing trees surviving the burn with "slightly scorched" crowns, an observation following earlier fires (Davis et al. 1968). In comparison, post-fire litter fall on the watershed burned by a high severity fire was minimal because most of the trees had either been killed by the wildfire or severely damaged and died within two years of the burn.

Salvage cutting and fuel reduction treatments implemented on the watershed burned by a high severity fire impacted the post-fire loadings of flammable fuels. The fraction represented by standing trees was (essentially) eliminated on this watershed by the treatments. However, stem sections, branches, and twigs on the forest floor increased significantly as a result of these interventions. Therefore, while the probability of future crown fire on the watershed has been reduced, the possibility of surface fire remains.

CONCLUSIONS

Recovery of the Stermer Ridge Watersheds from the Rodeo-Chediski Wildfire to date is related to the fire severities that the two watersheds experienced. The watershed burned by a high severity fire underwent a conversion from a southwestern ponderosa pine forest ecosystem to a landscape of grasses, forbs, and a few shrubs. This post-fire cover of plants is likely to persist in varying forms in the absence of artificial tree regeneration. Recovery of the hydrologic functioning on this watershed has begun on a limited scale with elimination of fire-induced water repellent soils and resulting decrease in the magnitudes of overland flows of water; a reduction in soil erosion and deposition and, therefore, sediment accumulations in the stream channel; and a gradual but continuous return of peak stormflows to pre-fire levels. It is anticipated, however, that the overall hydrologic functioning of this watershed will not approach pre-fire conditions for many years. Flammable fuels represented by standing trees have been eliminated as a result of the salvage cutting and fuel reduction treatments, but there has been an increased in the fraction comprised of stem sections, branches, and twigs on the forest floor. Therefore, while the possibility of a future crown fire has declined, the potential for surface fire remains.

Much of the watershed exposed to a low severity fire was either unburned or burned by surface fire at a low severity. The sites that were burned are recovering slowly from the Rodeo-Chediski Wildfire, a progression that is likely to continue into the future in the absence of further wildfire events or other major disturbance. While small areas of this watershed were burned at higher fire severities, these areas were isolated and scattered on the landscape. Much of the hydrologic functioning of the watershed is also returning to its pre-fire level. Post-fire loadings of flammable fuels were largely unchanged from their pre-fire estimates. This watershed remains vulnerable to future wildfire events as a consequence.

MANAGEMENT IMPLICATIONS

Information presented in this paper should be helpful to managers in anticipating the impacts of future wildfire events of varying burning severities on the ecosystem resources, hydrologic functioning, and flammable fuels in southwestern ponderosa pine forests. Duration of these post-fire impacts into the future might also be estimated. Identifying the need for implementing rehabilitation measures to mitigate the impacts when necessary could also possible in some instances.

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FUNCTION OF RIPARIAN VEGETATION IN RETAINING SEDIMENT IN WATERSHEDS

Daniel G. Neary,[1] Silke Buschmann,[2] and Peter F. Ffolliott[3]

Riparian areas are special vegetation zones that are frequently used as buffer strips to mitigate sediment movement from upland forest and agricultural management areas (Neary et al. 2010). These areas are often called streamside management zones. Understanding any concept and accompanying literature is ultimately based on knowledge of the descriptive terminology used. Therefore it is appropriate to discuss the terms used in the literature to describe since there is a lack of uniformity in how these near-stream landscape units are named. The most commonly used terms mentioned by Neary et al. (2010) are "riparian zones", "riparian buffers", "riparian management zones", "buffer strips", "filter strips", "wetlands", "greenways", "grassed waterways", and "streamside management zones". Various combinations of economic, ecological, and regulatory factors govern their size, shape, function, importance, and management (Williams et al. 2003).

The term "riparian zones" has variants in "riparian buffers", and "riparian management zones" that imply the same concept. Riparian zones are transition zones between upland and aquatic ecosystems that are under the influence of shallow groundwater (Mitsch and Gosselink 2007). Technically, riparian zones are those that meet the definition of "riparian". The riparian zone of a river, stream, or other body of water is the land adjacent to that body of water that is, at least periodically, influenced by flooding. Riparian zones are usually highly productive ecotones or transition zones between upland and aquatic ecosystems that are under the influence of flooding and shallow groundwater. They can vary from a few meters wide in steep terrain to hundreds of meters in coastal plains. Often, the term "riparian management zone" is used to broaden the concept beyond that of a buffer (Lowrance et al. 1985). Riparian management zones and buffers refer to management-designated areas that may or may not include all the components of the riparian zone and can also include some upland areas.

The transition between two different land uses where one land use, the buffer, mitigates the effects of the other is referred to as a "buffer strip" (Karr and Schlosser 1978, Comerford et al. 1992). This transition zone allows runoff and its pollutant load to be reduced before reaching surface waters by filtering, deposition, infiltration, adsorption, uptake, and decay. The primary function of "filter strips" is on the filtering pollutants that leave upland portions of landscapes. Filter strips are usually viewed as zones to remove sediment from runoff (Cooper et al. 1987), but they will also filter out nutrients like nitrate nitrogen if they are wide enough (Lowrance et al. 1984). Filter strips are most often used at field edges but they can also function well along streams.

"Greenways and grassed waterways" are specialized grass or herbaceous plant zones along first- and second-order drainage areas in agricultural landscapes (USDA Agriculture Research Service 1987). They are sometimes used in forested areas but not frequently. In general, grassed waterways typically carry concentrated flows but have a high potential for reducing runoff volume and velocity, sediment transport, and chemicals coming from agricultural watersheds through sediment detention and water infiltration processes (Fiener and Auerswald 2003).

"Wetlands" are often used as an interchangeable term with the word riparian. This landform includes fifteen different, popular terms such as bog, fen, marsh, moor, swamp, bottomland, etc. (Mitsch and Gosselink (2007). Although it also describes land that is transitional between terrestrial and aquatic ecosystems, there are a number of distinctions which really set wetlands apart from other landscape units. Cowardin et al. (1979), in their classification of USA wetlands, defined the term as referring to lands where the water table is at or near the surface or the land is covered by shallow water. For the classification system that they used, wetlands have the following distinct attributes: (1) they support hydrophytic plants, (2) they contain predominantly hydric soils, and (3) they are water saturated or covered by water during part of the growing season. In the complete absence of active land management (e.g. wilderness, National

1 Rocky Mountain Research Station, US Forest Service, Flagstaff, Arizona
2 School of Forestry, Northern Arizona University, Flagstaff, Arizona
3 School of Natural Resources and the Environment, University of Arizona, Tucson, Arizona

Parks, special reserves, etc.), wetlands and riparian areas provide the functions that SMZs do in managed landscapes.

Streamside management zones can include riparian zones as well as upland areas (Phillips et al. 1999). The term encompasses all potential functions and management objectives for landscape units adjacent to streams. Thus, it is not tied to the hydrologically functional area of the riparian zone since it can include parts of upland areas, and it includes functions other than buffering. For these reasons, it is a preferable term to define the managed landscape units along stream courses.

Riparian zones have many functions. These include wildlife habitat, water source, recreation areas, climate moderation, and water quality protection. They are often recognized as important barriers or treatment areas that protect water resources quality from non-point source pollution sediment. The vegetation and the geomorphic characteristics of these buffer strips produce infiltration, filtering, and deposition of sediment-laden water flowing off of intensively managed forest and agriculture lands. The effectiveness of riparian vegetation in trapping sediment depends upon the velocity of water flow, size distribution of sediments, slope and length of slope above the riparian buffer, slope and length of the buffer strip, depth of water flow into the riparian zone, vegetation characteristics such as type, density, and height. Data on sediment removal by forest vegetation buffer strips suggest that two main actions occur. First, the forest edge environment promotes sediment removal from surface runoff. Second, the sediment is sorted as it moves through lower gradient zones of the riparian buffer. This paper examines these processes and illustrates them with examples from forest management and agriculture.

SEDIMENT RETENTION PROCESSES

Surface Roughness

Surface roughness features include coarse-woody debris, live woody and herbaceous vegetation, emergent wetland vegetation, the soil organic horizon (litter), boulders, rock outcroppings, surface depressions, and land that slope away from the stream edge. In an agroforestry landscape, streamside management zones usually increase the surface roughness compared to adjacent tilled fields and pasture, which in turn reduces runoff velocity and thus enhances deposition of sediment particles and increases the opportunity for runoff to infiltrate into soils (Neary et al. 2010). Sufficient contact time between water, vegetation and soil is important to obtain a decrease in runoff velocity. Some of the other factors that interact with surface roughness include the width of the buffer zone, vegetation type (density, stiffness and height), and slope. These characteristics ultimately affect the trapping efficiency (Schultz et al. 2009).

Johnson and Buffler (2008) proposed a simple but useful classification system for surface roughness based on easily measurable or observable features (Table 1). The initial screening factor for their classification system is the percentage of the streamside management zone that contains recognizable surface roughness features. Examples of the three categories (low, moderate, and high). The Johnson and Buffler (2008) paper recognizes that surface roughness is not necessarily uniform and can be recognized and mapped. Figure 1 shows an example of an actual streamside management zone with sections characterized by variable surface roughness. This feature of can be fairly uniform or highly variable.

Table 1: Streamside Management Zone surface roughness classification system of Johnson and Buffler (2008).

Degree of Surface Roughness	Description
Low	• Less than 35% of the land surface contains surface roughness features. • SMZs with exposed mineral soils as a result of human use automatically have a low degree of surface roughness, as do managed areas (e.g. areas that are intensively grazed, mowed, or used for agriculture).
Moderate	• Between 35 and 65 percent of the land surface contains surface roughness features • For an open (non-forested) system, such as shrub-steppe or wet meadow, it must not be intensively grazed, mowed, hayed, or intensively managed • Usually, there will be clumps of woody vegetation establishing due to lack of mowing • Vegetation must be rough and dense. • There must be surface roughness features other than herbaceous vegetation, such as woody debris, boulders, or hummocky topography, over at least 5% of the land surface by aerial coverage. • The surface organic horizon (duff layer) is intact throughout the buffer unit.
High	• Greater than 65 % of the land surface contains surface roughness features. • The microtopography is complex with undulating topography resulting from previous geologic and hydrologic events. The land surface does not slope smoothly and consistently toward the stream. • The SMZ unit is forested or covered with dense stands of riparian scrub, shrub-steppe vegetation, or dense rush/sedge vegetation. • The surface organic horizon (duff layer) is intact throughout the buffer unit. Natural occurrences of mineral soils, such as tip-ups (trees that fall over bringing the root crown and attached mineral soils to the soil surface), may be present. In natural shrub-steppe plant communities, 35 percent exposed mineral soil is common and undisturbed sites may be covered with cryptogamic crust. • In forested areas, dead-and-down wood and rotting logs and stumps are common. Specifically, coarse woody debris (>25 mm.) is scattered about and older woody debris is being incorporated into the organic horizon. • There is a well-developed grass and forb layer. However, in shaded SMZ woodlands this will not always be the case. A dense grass-forb layer is not necessary, although it remains important for a high degree of surface roughness as long as other factors are present. • Boulders and exposed bedrock are common and, where present, add microtopographical complexity. This feature is not required.

Sediment Size

Often one of the main functions expected of riparian vegetation is to reduce sediment movement in runoff. A riparian buffer is meant to function as a physical filter by removing sediment that is being transported by water entering the zone. The effectiveness of vegetated areas in trapping sediments depends on a number of factors, the principal one being water velocity (Karr and Schlosser 1978). The size of sediment transported into and through streamside management zones is a function of water velocity, which is in turn affected by surface roughness (Comerford et al. 1992). Competance (see Equation 1) is the maximum size of sediment particles moved by water and is proportional to the sixth power of water velocity (Hewlett and Nutter 1969):

Equation 1: $C = kV6$

where:

C = Competance

k = coefficient

V = water velocity

Small reductions in velocity can make drastic changes in the sizes of sediment particles carried in surface runoff. Sand-sized fractions of suspended sediment will usually deposit quite rapidly as velocity drops off (Figure 2). Silt-sized sediment is able to settle out if runoff water is detained in surface roughness and depression and velocities are dropped to near or <1 m sec-1 (Hjulström 1939). Clay-sized sediment fractions do not readily settle out of a water column in even still water. The wider the buffer and the higher the surface roughness, the more likely that sediment trapping efficiencies will be high (Dosskey et al. 2008).

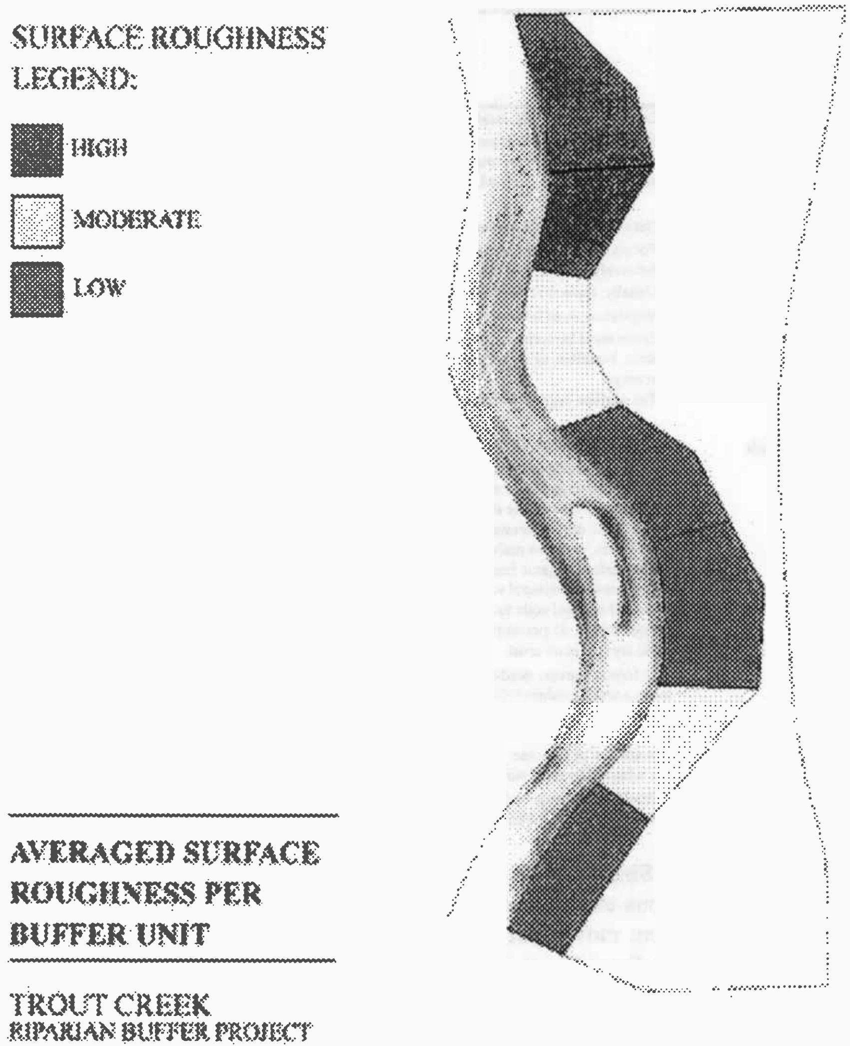

Figure 1. Surface roughness map for the Trout Creek riparian buffer streamside management zone (Adapted from Johnson and Buffler 2008).

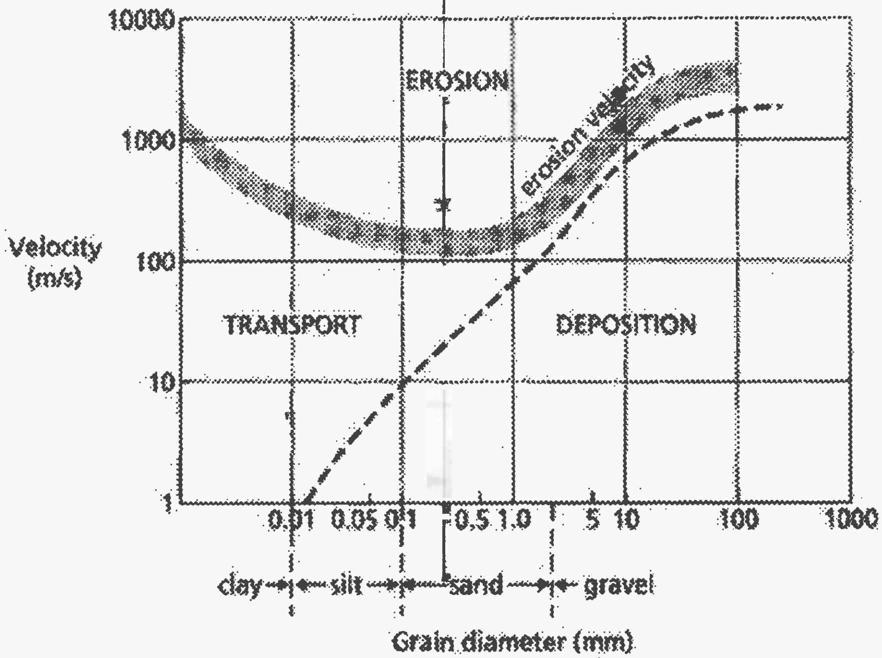

Figure 2. Relationship of suspended sediment size and transport velocity to erosion, transport, and deposition (Adapted from Hjulström 1939).

Water Velocity

Water velocity is an important concept influencing for sediment detention in riparian areas. Sediment transport in water is a function of water velocity and described in detail by Yalin (1977) and Haan et al. (1994). Any feature that reduces the velocity of water flowing into the buffer area will increase deposition of sediment. It will also provide additional time for infiltration of upland runoff into the soil, facilitating nutrient uptake and transformations (Schultz et al. 2009). Excessive water depths, especially in areas of concentrated flow, will reduce filtration functions since the water column will not interact with surface features. The sediment –laden runoff can then move directly into stream channels (Dosskey et al. 2002).

Slope Uniformity

Streamside management zones are often viewed to function by receiving relatively uniform sheet flow from uplands, and detaining the sediment and other pollutant loads carried in upslope runoff. However, hillslope geometry may force surface runoff into concentration points, increasing the velocity and depth of water flow into adjacent riparian areas. These concentrated flows along natural or artificial drainage lines act more like point source inputs to streams and lakes rather than nonpoint ones. Roads frequently cause this sort of hillslope water flow pattern, but rock outcrops and coarse woody debris can produce the same effect. Streamside management zones are able to transform concentrated flow to sheet flow if the receiving area is wide enough, surface roughness is high, and drainage pathways are not well-defined. This situation is the most common cause of low buffer effectiveness since it routes upslope runoff straight through any streamside management zone, minimizing any filtering of pollutant-laden water. Sediment moving in surface runoff moves straight through the buffer rather than depositing within depressions (Figure 3).

The function of a streamside management zone in improving water quality depends on the assumption of an essentially uniform flow of water (above and below ground) through the zone. Channelized or concentrated flow at the surface or in the soil will reduce the water quality improvement effectiveness. In some landscapes, flow tends to be dominated by channelized flow towards the base of slopes (Nutter

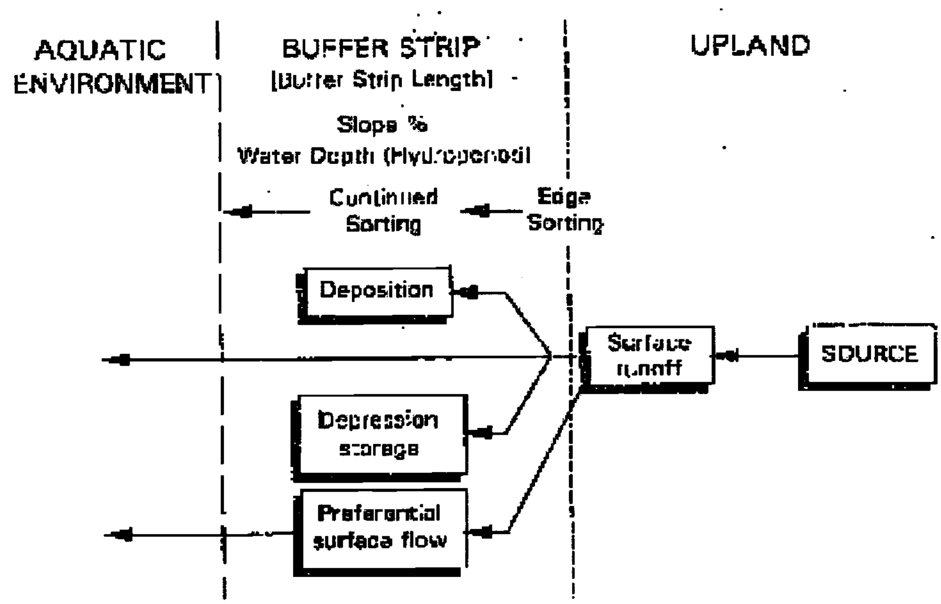

Figure 3. Schematic of sediment movement through a streamside management zone (From Comerford et al. 1992).

and Gaskin 1988). Rapid movement of water in ephemeral drainages and large soil macropores can quickly move sediments, nutrients, and pesticides across and through the riparian area and into stream channels. This reduces the time available for degradation and retention processes to occur. Flow routed into and through streamside management zones preferentially as concentrated surface runoff aggravates this situation.

Adjacent Slopes

Excessive slope can limit the effectiveness of riparian buffers. The steeper the slope the higher the velocity of surface runoff and, therefore, the greater the risk of concentrated flows breaking through an SMZ. As will be seen in the next section of this paper, most models for determining streamside management zone size factor in slope. Some researchers have argued the case that stream buffers should increase exponentially with slope (Nieswand et al. 1990) while others found that the width increase with increasing slope should be linear (Trimble and Sartz 1957, Swift 1986). There is a slope point at which buffers cease to be effective at all in containing runoff from adjacent uplands. Recommendations for this breakpoint range from 10 to 40% (Cohen et al. 1987, Nieswand et al. 1990, and Wenger 1999).

Riparian Buffer Widths

The widths of streamside management zones reported in the literature are quite varied. There is no one-size-fits-all although many are in the 10 to 15 m range. The important feature of width is that it is flexible enough to meet the objectives functions of land managers (Minnesota Forest Resources Council 2005). Streamside management zone width is important since it allows sufficient time for water draining off uplands to contact vegetation, litter, and soil and decrease velocity, allowing for deposition of sediment. Width is important also in providing sufficient space for upland runoff, which might be entering as concentrated flow, to convert to sheet flow (Schultz et al. 2009).

Flow Conditions and Floods

If a riparian vegetation zone is not covered by water during most of the year, water depth and flow are minor considerations. However, if it is primarily wetland and frequently saturated, then the timing of surface runoff could influence the effectiveness of sediment filtration for water quality purposes. If high flows or concentrated flows reduce contact time and interaction with surface roughness features, the beneficial effect of vegetation, litter, and coarse woody debris is greatly reduced (Comerford et al. 1992).

Vegetation Characteristics

Key vegetation characteristics that contribute to the effectiveness of riparian areas in retaining sediments entrained in runoff include the density, height, and lateral distribution of understory plants as well as the amount of coarse woody debris. Both contribute to the surface roughness characteristics of riparian zones. Greater sediment filtering is normally found with wide riparian zones typical of the larger river systems like the Colorado River, Salt River, Gila River, San Pedro River, and Rio Grande (Comerford et al. 1992). However, even these rivers have some reaches where riparian vegetation thins down to narrow strips (Baker et al. 2003).

LITERATURE EXAMPLES

Sediment is an important water quality parameter, since higher than normal levels can harm aquatic organisms and habitats, and render water unacceptable for domestic or recreation purposes. Sediment yields after forest harvesting are highly variable depending on such factors as soils, climate, topography, ground cover, and watershed condition. Although sediment yields often increase after harvesting due to the physical disturbance that exposes soil to erosion processes, these increases are usually transient due to vegetation re-growth, and are mitigated by slash and litter retention and other aspects of surface roughness. However, the duration of sediment increases above pre-disturbance conditions can vary considerably. Beschta (1978) reported effects lasting 6 years after logging while Lynch and Corbett (1990) saw effects out to ten years. Neither study lasted long enough to document recovery to pre-disturbance conditions.

The largest increases in sedimentation documented in the literature have been associated with post-harvest mechanical site preparation in the absence of streamside management zones (Beasley 1979), or with slope instability (O'Loughlin and Pearce 1976), road construction (Swanson et al. 1986), or highly erosive soils (Beasley and Granillo 1988). Best management practices are most effective on sediment when properly planned and implemented prior to, during, and after harvesting. Most of these guidelines relate to designing, constructing, and maintaining major access roads, logging roads, skid trails, and landings (Binkley and Brown 1993). These areas are the primary sources for 90 percent of the sediment generated by harvesting (Reid and Dunne 1984, Swift 1986). The underlying principles of BMP guidelines in this context are to minimize disturbances in riparian areas, reduce the erosive power of runoff from bare road surfaces, and to maintain the normally high infiltration capacity of forest soils.

Sediment movement to streams is an on-going environmental concern in managed forest watersheds, but it also occurs naturally without active management. Watersheds vary greatly in their natural sediment load characteristics. Both natural and anthropogenic sediment deposits can be re-entrained after initial deposition in ephemeral or perennial stream channels, and move downstream for long time periods (>100 years) and distances. The cumulative effects of erosion and sedimentation that occurred centuries ago from agriculture or forestry can present land managers with many challenges such as channel bank collapse, tunnel erosion, and channel sediment transport to sensitive water resources (Neary 2002, DeBano et al. 2005).

Erosion Rates

Reference sediment-yield baselines have been discussed by Neary and Michael (1996), Neary (2002), and DeBano et al. (2005). Natural erosion rates (geologic erosion) for undisturbed forests in the western United States of <0.01 to 5.53 Mg $ha^{-1}yr^{-1}$ are generally higher than eastern United States yields of 0.1 to 0.2 Mg ha^{-1} yr^{-1}, but do not approach the upper limit of geologic erosion (15 Mg ha^{-1} yr^{-1}, Schumm and Harvey 1982). Australia's geologic erosion rates range from 0.04 to 0.86 Mg ha^{-1} yr^{-1} (Wasson et al. 1996). The measured differences at both continental and local scales are due to natural site factors such as soil and geologic erosivity, rates of geologic uplift, tectonic activity, slope, rainfall amount and intensity, vegetation density and percent cover, and fire frequency. Landscape-disturbing activities such as mechanical site preparation (15 Mg ha^{-1} yr^{-1}; Neary and Hornbeck 1994), agriculture (560 Mg ha^{-1} yr–1; Larson et al. 1983), and road construction (140 Mg ha^{-1} yr^{-1}; Swift 1986, Binkley and Brown 1993) produce the most sediment loss and can match or exceed the upper rate of geologic erosion. Erosion rates can be as high as 1,000 Mg ha^{-1} yr^{-1} in small gullied basins, and along with channel banks they are the main sources of sediment for Australian rivers (Wasson 1994, Prosser et al. 2001). Roads, including tree harvesting access roads and skid tracks, are particularly problematic and chronic contributors of sediments to streams.

Sediment Yield Variations

Variation in suspended sediment concentration can be quite large both within and between individual catchments and regions (Binkley and Brown 1993). Most undisturbed forested catchments have suspended solid concentrations <5.0 mg L^{-1} and stormflow peaks >100 mg L^{-1}, but some routinely average higher than 20 mg L^{-1}. Storm runoff from steep watersheds with highly erosive soils can have average suspended solids concentrations >400 mg L^{-1} (Beasley 1979). Responses of forested watersheds harvested without SMZs can be in the range of <2 to 43 times reference or pre-treatment conditions (Binkley and Brown 1993).

The effectiveness of SMZs in filtering out sediment from storm runoff has been demonstrated in a number of studies (e.g. Arthur et al. 1998, Dosskey 2001, Helmers et al. 2005, Loch et al. 1999, McKergow et al. 2003 and 2006). Sediment removals of 90% or more have been measured, and reductions in flow weighted mean suspended sediment concentrations of an order of magnitude have been documented (148 to 13 mg L^{-1}, McKergow et al. 2003). So, it is very clear that SMZs can work well in protecting or improving water quality. The big question is: "Can tree stands in SMZs be harvested without causing a significant deterioration in water quality and other important functions?" There are a limited number of published studies that address this question.

Minnesota

Hemstad et al. (2008) reported on fish habitat changes after several thinning treatments in a mixed hardwood forest with a SMZ. Reference areas had no tree felling at all. Riparian reference sites had upland areas that were clear-felled with a shortwood cut-to-length (CTL) system, but they retained a 30 m no-cut buffer zone (Mattson et al. 2000, Palik et al. 2000). Cut-to-length logging is a mechanized harvesting system in which trees are delimbed and cut to specified lengths (usually 3 m) directly at the tree stump. CTL is typically a two-machine operation with a harvester felling, delimbing, and cutting trees and a forwarder transporting the logs from the felling to a landing. The CTL riparian thinning treatment involved upland clear-felling and riparian zone harvesting to a residual basal area of 12.3 m^2 ha^{-1}. The final treatment was a whole tree-length (WTL) harvesting where the adjacent upland was clear-felled and the streamside management zone was thinned to the same residual basal area as the CTL treatment. In WTL logging trees are felled, delimbed, topped, and moved to a landing without being cut into smaller lengths. Hemstad et al. (2008) measured a small (15%) increase in streambed fine sediments and gravel embeddedness after harvesting. However, the sediment increase was catchment-wide in all stream reaches and involved the uncut reference reaches as well as the stream reaches that had SMZ harvesting. The effect was attributed to sediment washing off roads and stream crossings and into stream channels, not sediment derived from harvesting operations. This result points out the role of roads in producing much of the post-harvest sediment yield observed in forested catchments. Roads have been broadly recognized as important contributors to stream sediment loads (e.g. Beschta 1978, Swift 1986, Ziemer et al. 2000).

North Carolina

In another study, a coastal plain swamp (riparian zone) forest in North Carolina was clear-felled in May, 1998, leaving a 10 m wide uncut portion of the riparian zone as a buffer (Ensign and Mallin 2001). The harvested area was along the lower end of a 479 km^2 catchment. A significant increase in total suspended solids was measured in the harvested catchment during rainfall in June, July, and August after harvesting. Compared to a nearby uncut reference catchment, suspended solids were consistently 3 to 10 mg L^{-1} higher in the logged catchment during that 3-month period. One peak of 111 Nephelometric Turbidity Units (NTUs) was measured in the treated catchment during the first month after clear-felling. In contrast, extreme rainfall during Hurricane Bonnie in late August 1998 did not produce a notable effect on total suspended solids in either catchment. Stream samples collected 8 days after the hurricane landfall and 85 mm of rain did not show any notable impact to total suspended solids as a result of the storm. Turbidity in the harvested area was not significantly different from the uncut reference catchment although it did rise from 6 to 14 NTUs due to the hurricane. This was not a significant rise in NTU level, but it demonstrates that harvesting in wide riparian zones can lead to at least small and transient increases in turbidity even if BMPs exclude SMZ harvesting.

Malaysia

Gomi et al. (2006) studied the effects of logging lowland tropical rainforest in Butik Tarek Experimental Watershed in Malaysia that included 20 m riparian buffers. Catchments (14 to 38 ha) included a reference site with no harvest (C1), a 20 m buffer (C2), a 20 m buffer with a high road density and in close proximity to the riparian area (C2T), and a riparian partial harvest (C3). Catchment areas outside the streamside management zone were clearcut and logs extracted by skidders and log trucks. There was no significant difference in sediment delivery to the Butik Tarek stream in catchments C1 and C2. Catchments C2T and C3 produced 6 and 5 times the sediment volume into channels, respectively, than the C1 catchment. Roads and skid trails in C2T and C3 with steep gradients (>20%) and a high degree of connectivity to the catchment channels were responsible for the high sediment delivery to the Butik Tarek stream, not the tree-felling operation per se, as observed, for example, by Sidle et al. (2004).

Mississippi

Logging with and without streamside management zones in twelve mixed hardwood forest catchments with highly erodible loess soils in Mississippi, USA, was evaluated by Keim and Schoenholtz (1999). Treatments consisted of: 1) unrestricted cable- and skidder-harvesting including the streamside management zone, 2) cable-yarding only harvesting in a streamside management zone of 30 m width, and unrestricted cable- and skidder-harvesting of outside the streamside management zones, 3) no-harvesting within a 30 m-wide streamside management zone and unrestricted cable- and skidder-harvesting of other areas, and 4) a no harvesting reference watershed. Streams in catchments without any streamside protection had mean total suspended solids (TSS) concentrations 2.9, 3.2, and 1.8 times the concentration of the reference watershed stream with a buffer and no harvesting (mean TSS of 244.2, 272.0, 147.4, and 83.7 mg L^{-1} for treatments 1, 2, 3, and 4, respectively). While this indicated the water quality protection value of streamside management zones, the responses of treatments 2 and 3 indicated the importance of minimizing soil disturbance within riparian areas. The results suggest that the streamside management zones did not function in trapping sediment originating in harvested areas outside the buffer. Instead, the reduced sediment input into streams was achieved by creating less soil disturbance within the streamside management zone. The authors concluded that riparian zone prescriptions should focus on eliminating machine traffic within 10 m of streams.

Tasmania, Australia

A study was conducted in northwest Tasmania, Australia, to evaluate the water quality benefits of establishing a riparian streamside management zone and the effects of tree harvesting in this zone (Neary et al. 2010) This case study consisted of a 20-year-old Eucalyptus nitens pulpwood plantation in a streamside management zone of an intermittent stream that was harvested according to the Tasmania Code of Forest Practice. A machinery exclusion zone immediately adjacent to the stream limited machinery traffic. Ground cover and water quality pre- and post-harvesting were measured to identify the major sources of sediment in this headwater catchment, and to determine the effect of tree harvesting. Tree harvesting in the Tasmanian study resulted in minimal mineral soil exposure and increased surface roughness. Post-harvesting turbidity levels in streamflow were similar to pre-harvest levels (<2.5 Nephelometric Turbidity Units (NTUs) exiting the catchment) (Figure 4). Much more significant sources of sediment were a road, a dam that was accessible to cattle, and a cultivated paddock. These sources led to turbidities of c. 300 NTUs in a dam immediately below these points and above the harvested stream reach during a storm in late June 2009. In-stream dams installed many years earlier to store water for stock and irrigation, acted as very effective sediment traps. The riparian buffers and other BMPs used in these landscapes were effective at protecting water quality. This demonstrates that forest-harvesting operations can be conducted in riparian zones without increasing stream turbidity, if existing BMPs are followed.

Slope and Riparian Buffer Size

Helmers et al. (2005) reported on the mean sediment reduction with riparian vegetation zones of <5 to 20 m and slopes of 2 to 16% (Table 2). Mean sediment reduction was best with over 10 m in width. However, even riparian zones <5 m in width can be effective at trapping over 77% of sediment inputs.

Table 2: Mean sediment reduction on different slopes with riparian buffer widths of <5 to >20 m (Helmers et al. 2005).

Buffer Width	Slope Range		Mean Sediment Reduction	Sediment Reduction Range	
	Minimum	Maximum		Minimum	Maximum
m	%	%	%	%	%
<5	2	16	77.5	45	97
5 to 10	2	16	83.3	57	100
10 to 15	2	16	96.3	84	100
15 to 20	6	16	96.2	85	100

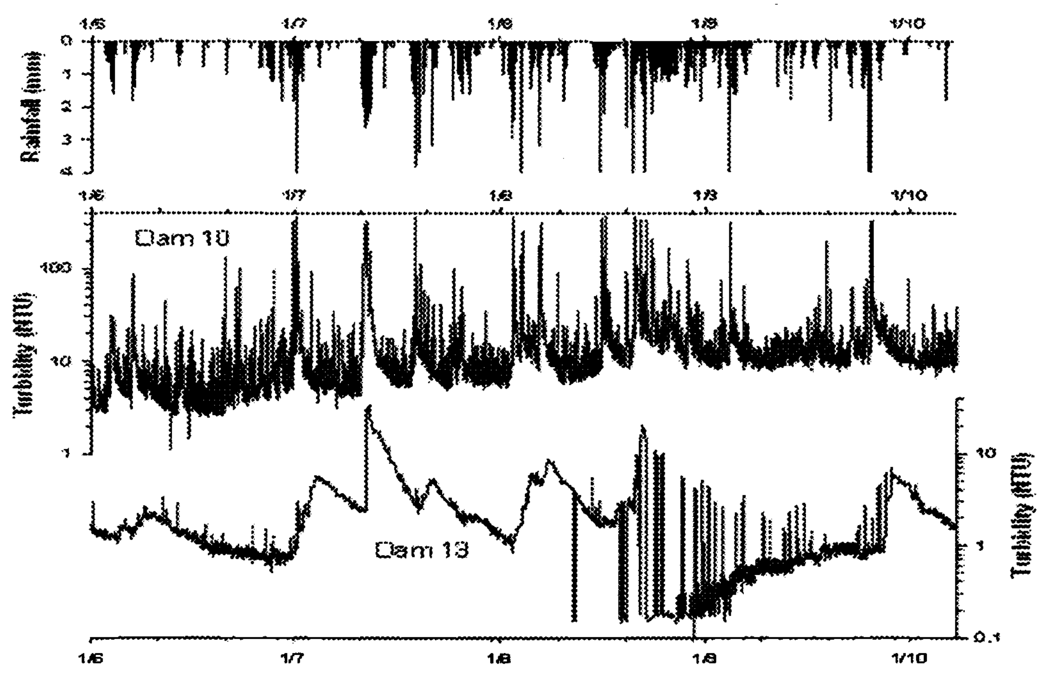

Figure 4. Rainfall in mm on the Pet River tributary catchment during the 2009 winter rainy period, and continuous turbidity measurements at D10 and D13 in mm per 15-minute intervals from 27 May 2005 to 9 October 2009. Y-axis units for Dams 10 and 13 are in NTUs on a log scale (From Neary et al. 2010).

WHERE RIPARIAN VEGETATION BUFFERS DON'T WORK

There are situations where riparian buffers do not function well in trapping sediment derived from adjacent upland landscapes (Comerford et al. 1992). These involve processes that produce focused, directional routing of sediment-laden water into and through riparian zones, or situations where the normal sediment detention mechanisms are not functional.

The chief causes are roads, non-uniform slopes, flood inflows, scarce vegetation, and channel sediment sources. Some of these causes are unique and others interact to produce cumulative sediment loss effects.

Roads

Roads are the leading cause of runoff that breaks down the filtering function of riparian vegetation (Swift 1986). If roads are not designed properly, runoff can concentrate on road surfaces and deliver large volumes of water directly into streams. The Variable Source Area Concept (Hewlett and Hibbert 1967) describes how the perennial channel system expands during precipitation as areas at the head of and adjacent to perennial channels become saturated during storm events. Roads result in the expansion of

effective drainage network in a catchment. High levels of soil disturbance (e.g., physical exposure, removal, site preparation, and road construction) can shift more water originating from precipitation into overland flows and surface stormflows that produce damaging stream flows and concentrated flows through riparian vegetation and directly into stream channels (Neary 2002). High velocity water flows detach road surface sediments and lead to road fill and side slope gully erosion. Over-dense road networks also contribute to this problem. Gomi et al. (2006) reported that tree harvesting in Malaysia using a 20 m vegetation buffer increased stream sediment depositions by 3-fold, but adding roads into the mix increased sediment by an additional factor of 6.

Non-Uniform Slopes

This condition was discussed previously as a contributing factor to reduced sediment filtration in riparian zones. It sets up conditions for rapid movement of water into and through riparian buffers (Comerford et al. 1992). Rapid movement of large volumes of water into the riparian zone buffer reduces the amount of sediment deposited within the riparian zone or in depressions (Figure 3).

High Rainfall and Flood Inflows to Riparian Areas

High amounts of rainfall and subsequent runoff can have the same effect on sediment filtering as concentrated flows from roads or non-uniform slopes (Comerford et al. 1992). Substantial amounts of coarse and fine sediments carried in flood flows are often deposited in riparian areas during these inundation events, but significant amounts are also moved through riparian systems either further downstream or into adjacent uplands. Riparian vegetation is able to trap sediments up to a variable failure point determined by water flow velocity and sediment load (Dosskey et al. 2002).

Channel Sediment

Some channel sediments can be temporarily stabilized by riparian herbaceous and woody vegetation. However, fluvial dynamics during flood events can easily remobilize these sediments (Baker et al. 2003). Vegetation is often viewed as a good mechanism for stabilizing bank and first terrace sediments (Medina et al. 1996). The main exception to this rule is the presence of vertical banks due to channel degradation (Neary et al. 2001). Vertical banks are major sources of sediment and will only be stabilized by vegetation when the angle of repose of the banks declines below a stability angle (<45°) determined by the sediment texture and moisture content (Bell 1998). Bank collapse occurs as a result of streamflow undercutting or animal activity.

SUMMARY AND CONCLUSIONS

Riparian areas are special vegetation zones that are frequently used as buffer strips to mitigate sediment movement from upland forest and agricultural management areas. These areas are often called streamside management zones. They are strips of land along rivers or lakes that are given special management consideration. Various combinations of economic, ecological, and regulatory factors govern their size, shape, and management. Riparian zones function as and are often recognized as important barriers or treatment areas that protect water resources from non-point source sediment. Vegetation and the geomorphic characteristics of these buffer strips produces infiltration, filtering, and deposition of sediment-laden water flowing off of intensively managed forestry and agriculture lands. The effectiveness of vegetation in riparian areas for trapping sediment depends upon the velocity of water flow, size distribution of sediments, slope and length of slope above the riparian buffer, slope and length of the buffer strip, depth of water flow into the riparian zone, and vegetation characteristics such as type, density, and height. Data on sediment removal by forest vegetation buffer strips suggests that two main actions occur. First, the forest edge environment promotes sediment removal from surface runoff. Second, the sediment is sorted as it moves through lower gradient zones of the riparian buffer. This paper examined these processes and illustrated them with examples from forest management operations and agriculture. In most instances, vegetation is quite effective in detaining sediments derived from uplands. However, the use of Best Management Practices on forest and grassland watersheds is really the first "line of defense" in reducing sediment input into streams. There are situations where even vegetation buffer strips can't cope with sediment movements. These situations are poorly designed roads, non-uniform slopes, flood flows, and channel sediments, especially vertical banks.

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